An Assessment of the Health Implications of Aviation Emissions Regulations

by

Christopher J. Sequeira

S.B. Aeronautics and Astronautics
Massachusetts Institute of Technology, 2005

SUBMITTED TO THE DEPARTMENT OF AERONAUTICS AND ASTRONAUTICS
AND THE ENGINEERING SYSTEMS DIVISION
IN PARTIAL FULFILLMENT OF THE REQUIREMENTS FOR THE DEGREES OF

MASTER OF SCIENCE IN AERONAUTICS AND ASTRONAUTICS
AND MASTER OF SCIENCE IN TECHNOLOGY AND POLICY
AT THE MASSACHUSETTS INSTITUTE OF TECHNOLOGY

FEBRUARY 2008

Signature of Author _____
Department of Aeronautics and Astronautics and Engineering Systems Division
January 16, 2008

Certified by _____
Ian A. Waitz
Jerome C. Hunsaker Professor of Aeronautics and Astronautics
Thesis Supervisor

Accepted by _____
Dava J. Newman
Professor of Aeronautics and Astronautics and Engineering Systems
Director, Technology and Policy Program

Accepted by _____
David L. Darmofal
Associate Professor of Aeronautics and Astronautics
Chair, Committee on Graduate Students

An Assessment of the Health Implications of Aviation Emissions Regulations

by

Christopher J. Sequeira

S.B. Aeronautics and Astronautics
Massachusetts Institute of Technology, 2005

Submitted to the Department of Aeronautics and Astronautics and the Engineering Systems Division on January 16, 2008 in Partial Fulfillment of the Requirements for the Degrees of Master of Science in Aeronautics and Astronautics and Master of Science in Technology and Policy at the Massachusetts Institute of Technology

Abstract

An exploration of the health implications of aviation emissions regulations is made by assessing the results of a study of aviation's effects on United States air quality mandated by the Energy Policy Act of 2005. The Energy Policy Act study results estimated that aviation is responsible for 160 yearly incidences (with a 90% confidence interval of 64 to 270 incidences) of premature mortality of adults age 30 and over ($882 million in year 2001 dollars, with a 91% CI of $196 to $1830 million) due to exposure to particulate matter below 2.5 μm in size ($PM_{2.5}$) in the continental U.S. as reported by the Environmental Benefits Mapping and Analysis Program (BenMAP). Strong regional differences were noted; for instance, 18% of the total health incidences and costs occurred in Los Angeles County. Aviation was estimated to decrease ozone concentrations, causing small premature mortality disbenefits (health effects avoided due to the presence of aviation) of approximately 2 yearly premature mortality incidences ($9 million).

Primary particulate matter values in the Energy Policy Act study's emissions inventory had been generated using a conservatively biased version of the First Order Approximation method version 3.0 (FOA3), known as FOA3a, and the emissions of sulfur oxides (SO_x) had been incorrectly computed (underestimated by approximately 15%). To quantify the effects of these differences on health impacts, a comparison was made with a second inventory generated by CSSI, Inc. using FOA3. Based on the comparison, it is estimated that aviation was responsible for 140 to 160 yearly incidences of premature mortality from exposure to PM. 46% to 69% of the incidences were estimated to be due to changes in concentrations of ammonium sulfate secondary PM from SO_x, while ammonium nitrate secondary PM was estimated to be responsible for 18% to 20%. Concentrations of volatile primary PM from organic compounds and nonvolatile primary PM were responsible for 6% - 18% and 5% - 14% of the impact, respectively, while volatile primary PM from sulfates was responsible for 0% to 4%. Confidence intervals were not computed, and only the effects of changes in PM concentrations were assessed.

Based on the results, it is determined that changing regulations governing nitrogen oxide (NO_x) emissions and fuel sulfur content may be effective strategies to mitigate incidences of premature mortality due to aviation. An assessment was made of the effects of changing fuel sulfur concentration from 600 parts per million (ppm), as is typical of current jet fuel, to 15 ppm across the continental U.S. It is estimated that this change would reduce yearly premature mortality incidences due to aviation-related ambient PM exposure by 38%. Confidence intervals were not computed. The cumulative additional costs to refineries to produce 15-ppm fuel could be approximately $260 million, suggesting that the benefits may be comparable to the costs. However, such a strategy could have climate warming impacts since aviation sulfur emissions have a cooling influence on climate. It is also estimated that an immediate deployment of ultra-low sulfur fuel only for takeoffs from Los Angeles County could reduce aviation-related nationwide yearly incidences of mortality by 10%, with Los Angeles County health impacts being reduced by a factor of 2. The additional costs to refineries may be approximately $12 million, suggesting that such a policy may be cost-beneficial.

Finally, a brief exploration is done of a NO_x stringency assessment by the International Civil Aviation Organization's Forecasting and Economic Analysis Support Group (FESG), which predicted that an industry-wide investment of $30,000 - $40,000 would be required for every tonne of NO_x eliminated if the ICAO NO_x standard were to be increased by 10% in the year 2008. FESG found this to be the most cost-effective NO_x reduction strategy. A direct comparison with the Energy Policy Act and RSM results is difficult, yet an assessment finds that NO_x has health costs of only $2,000 per tonne in both sets of results.

Thesis Supervisor: Ian A. Waitz

Title: Jerome C. Hunsaker Professor of Aeronautics and Astronautics

Acknowledgements

When I arrived here in Boston at the age of 18 in August of 2001, I could not have imagined all of the ways in which I was to grow and change over the next six and a half years. I now look back and see that there are many people who have made my experiences possible and have been with me through good and bad times. First, I would like to thank my advisor Ian Waitz for giving me the opportunity to join PARTNER in the spring of 2006. I first experienced his great teaching in 2002 during Unified Engineering, and he has been an insightful and very supportive research and thesis advisor. I would also like to thank the students and postdocs that I've begun to know during my time in PARTNER: Anuja, Jim, Mina, Pearl, Andrew, Chris, Elza, Stephen, Steven, Doug, Tudor, and (now graduates) Gayle, Julien, and Karen. I appreciate all of them simply for being around, even when I was not sociable. Jennie has been excellent at supporting myself and every other member of PARTNER, and I want to thank her for that support. I'd also like to thank the Department of Aeronautics and Astronautics at MIT for offering me a graduate fellowship for the first year of my master's degree program; this allowed me to take the time that I needed to find a great research group.

I want to thank my close friends Amy and Lara for being a part of my life during their time at MIT and beyond. They have supported me through some very difficult times, and I am honored to know them. And finally, I wish to thank my parents Jennifer and Anselm and my sister Simone; without them, I most certainly would not be here or anywhere.

This work was supported by the Office of Environment and Energy, U.S. Federal Aviation Administration under Contract No. DTFAWA-05-D-00012, Task Order 0002 managed by Maryalice Locke, and by Task Order 0003 managed by Warren Gillette. Additional support was provided by Aerodyne Research, Inc. Subcontract No. ARI-10464-3, under University of Missouri-Rolla Subcontract No. 00016284-01, funded by prime sponsor FAA/National Academies/Transportation Research Board/Airport Cooperative Research Program Contract No. ACRP-02-04A. Any opinions, findings, and conclusions or recommendations expressed in this material are those of the author and do not necessarily reflect the views of the FAA, Aerodyne Research, Inc., University of Missouri-Rolla, or the National Academies Transportation Research Board Airport Cooperative Research Program.

Table of Contents

List of Tables

List of Figures

List of Abbreviations

ABL	Atmospheric boundary layer
AEDT	Aviation Environmental Design Tool
APU	Auxiliary power unit
ATADS	Air Traffic Activity Data System
BEIS	Biogenic Emissions Inventory System
BenMAP	Environmental Benefits Mapping and Analysis Program
BPR	Bypass ratio
BTS	Bureau of Transportation Statistics
CAEE	Committee on Aircraft Engine Emissions
CAEP	Committee on Aviation Environmental Protection
CAN	Committee on Aircraft Noise
CI	Confidence interval
CMAQ	Community Multiscale Air Quality Model
CMM	Cutanaceous malignant melanoma
COI	Cost of illness
CRF	Concentration-response function
dB(A)	A-weighted decibels
EI	Emission Index
EDMS	Emissions and Dispersion Modeling System
EPA	Environmental Protection Agency
EPAct	Energy Policy Act
ETMS	Enhanced Traffic Management System
FAA	Federal Aviation Administration
FESG	Forecasting and Economic Analysis Support Group
FID	Flame ionization detector
FOA3	First Order Approximation version 3.0
FOA3a	First Order Approximation version 3.0a
FRM	Federal Reference Method
GAV	Ground access vehicle

GSE	Ground support equipment
HAP	Hazardous air pollutant
HC	Hydrocarbon
ICAO	International Civil Aviation Organization
IMPROVE	Interagency Monitoring of Protected Visual Environments
kN	kiloNewtons
MM5	Mesoscale Model
MSA	Metropolitan Statistical Area
NAAQS	National Ambient Air Quality Standards
NASR	National Airspace System Resources
NEI	National Emissions Inventory
NMSC	Non-melanoma skin cancer
NO_x	Nitrogen oxides
ppb	Parts per billion
ppm	Parts per million
PM	Particulate matter
PM_{10}	Particulate matter 10 μm and smaller in size
$PM_{10\text{-}2.5}$	Particulate matter between 10 and 2.5 μm in size
$PM_{2.5}$	Particulate matter 2.5 μm and smaller in size
PR	Pressure ratio
RR	Relative risk
RRF	Relative reduction factor
RSM	Response Surface Model
SAGE	System for Assessing Aviation's Global Emissions
SANDWICH	Sulfate, adjusted nitrate, derived water, inferred carbonaceous material balance
SARP	Standard and Recommended Practice
SMAT	Speciated Model Attainment Test
SN	Smoke number
SO_x	Sulfur oxides
STN	Speciated Trends Network

ULS	Ultra-low sulfur
UV	Ultraviolet
VOC	Volatile organic compound
VSL	Value of Statistical Life
WTP	Willingness to pay

Introduction

Commercial aviation is a critical part of the world economy. It allows the rapid transport of material goods and people around the globe. Commercial aviation also generates jobs and fosters a wide variety of institutions in the aviation sector and beyond it. In the United States, the aviation industry is currently estimated to be responsible for 5.4 percent of gross domestic product (approximately $640 billion) and is also a strong contributor to exports (Joint Planning and Development Office 2007).

Aviation activity is projected to grow substantially over the next few decades. A study by a consortium of institutions modeled that the number of passengers flying per year could double to 4 billion worldwide by 2020 in a scenario of very high growth (European Commission on Constrained Scenarios on Aviation and Emissions 2005). Similarly, some forecasts indicate a doubling or even a tripling of demand for commercial air transport in the United States by 2025 (Joint Planning and Development Office 2007).

Some aviation emissions have negative impacts on the environment and on human health. In particular, aviation has adverse impacts on human health via perturbations to air quality (Ratliff, Sequeira, Waitz, et al. 2008). Although aviation activity is global, these air quality-related health effects often have strong regional variations. A study mandated by Section 753 of the Energy Policy Act of 2005 (United States Statutes at Large 2005) indicated that commercial aircraft emissions in 2005 increased ambient ozone concentrations by an average of 0.10 parts per billion (ppb) across the continental United States and increased ambient concentrations of particulate matter 2.5 μm and smaller in size ($PM_{2.5}$) by an average of 0.01 $\mu g/m^3$, based on an analysis using background air quality data in the U.S. Environmental Protection Agency's 2001 National Emissions Inventory (NEI). In the study, these air quality perturbations were estimated to have caused approximately 160 yearly incidences of premature mortality, with a 90% confidence interval of 64 to 270 incidences. 18% of these incidences occurred in Los Angeles County alone (Ratliff, Sequeira, Waitz, et al. 2008). 97% of the total mortality incidences were estimated to be caused by exposure to $PM_{2.5}$.

Aircraft emissions are regulated at national and international levels to mitigate the amount of pollutants entering the air from aviation activity. Current International Civil Aviation Organization (ICAO) and U.S. Environmental Protection Agency (EPA) regulations are based on measurements made at the engine exit plane (Code of Federal Regulations 2005, International Civil Aviation Organization 2005). There is also a voluntary standard in (ASTM International 2007) that specifies the maximum amount of sulfur in aviation fuel. The ultimate health impacts of emissions, however, depend on the dispersion and chemical transformation of pollutants and the number of people exposed to them. Air quality-related health impacts have regional variations partly due to these reasons, yet United States regulations prohibit states from setting their own aviation emissions standards (United States Code 2005d). Also, restrictions on aviation in the United States must be made without creating "an unreasonable burden on interstate or foreign commerce" (United States Code 2005e).

A variety of regulations exist that govern aviation emissions. However, to determine their ultimate effectiveness, one must determine the health effects of aviation emissions and how they are related to the regulations. Making such a determination requires the consideration of the dispersion and transformation of emissions and human exposure to the resulting concentrations.

Research questions

This thesis will attempt to explore several questions that address how aviation affects human health through changes in air quality. The first is, "what are the local and regional air quality-related health impacts of aviation, and how do they relate to aviation emissions?" Aviation activity produces a variety of emissions, and one must assess the dispersion, chemical transformation, and deposition of these emissions relative to the population distribution and baseline health incidences to determine the ultimate health effects.

The next question is, "how do current aviation emissions regulations relate to the health impacts of aviation?" Current regulations explicitly and implicitly address nonvolatile and volatile particulates as well as nitrogen oxides, sulfur oxides, and other gases. The final question that this thesis will explore is, "what alternatives might policymakers consider for modifying the current regulatory strategies?" Strategies have implementation costs, and these costs must be compared with the achievable benefits of implementing a strategy. This comparison also requires an awareness of the length of time needed for a strategy to become effective as well as the effects of the strategy on other areas such as public safety, climate change, and noise.

Organization of this thesis

This thesis is organized into three main chapters. Chapter 1 gives background on the emissions from aviation and their health effects, then discusses the Energy Policy Act study and analyzes its results. The chapter also highlights the conservatively biased assumptions and errors in the Energy Policy Act emissions inventory and makes a comparison with a second inventory by CSSI, Inc. (CSSI Inc. 2006), which was generated using less conservative assumptions and with some of the errors corrected. Both inventories are investigated to determine how various PM species contribute to health effects. Finally, the chapter explores the uncertainty in EPA's current methods of measuring ambient concentrations of particulate matter and apportioning mass among various PM species.

Chapter 2 gives background on current international and United States regulations that affect aviation emissions. This chapter also briefly discusses differences between U.S. and international regulations as set by ICAO. Finally, Chapter 3 assesses how emissions regulations influence aviation-related health effects and investigates the implications of a continental U.S.-wide switch to ultra-low sulfur (ULS) fuels. The chapter then explores the implications of a switch to ULS fuels just in Los Angeles County and concludes with a brief assessment of an analysis of NO_x stringency by ICAO's FESG (Forecasting and Economic Analysis Support Group) in the context of the health impacts predicted in Chapter 1.

1. Emissions from aviation and their related health effects

This chapter will give background on the various compounds emitted into the atmosphere by aircraft activity and other activities related to aviation. It will then discuss the health effects of those compounds and how they are determined. Next, the results of the Energy Policy Act study will be presented and analyzed to determine the health effects of various species of particulate matter at national and regional levels in the United States. The assumptions used to create the Energy Policy Act inventory as well as errors that exist in the inventory will be quantified through a comparison with another set of emissions data. Finally, the uncertainty in current particulate matter monitoring methods will be discussed.

1.1. Emissions from commercial aircraft

Gas turbine-powered aircraft emit a range of chemicals from their engines during operation. Approximately 70% of the emitted mass is carbon dioxide (CO_2), and 30% of the mass is water (H_2O) (Federal Aviation Administration 2005). Less than 1% of the emitted mass from aircraft engines consists of nitrogen oxides (NO_x), carbon monoxide (CO), sulfur oxides (SO_x), unburned hydrocarbons (HCs), and small particles (known as particulate matter, or PM). Engine emissions also consist of trace compounds like bits of metal from engine abrasion as well as ions and radicals with a very short lifetime (Kugele, Jelinek and Gaffal 2005). A small subset of HCs (specifically, some volatile organic compounds or VOCs) and PM can cause cancer or other serious health effects such as birth defects. These toxic chemicals are referred to by the United States Environmental Protection Agency (EPA) as hazardous air pollutants (HAPs) (U.S. Environmental Protection Agency 2002). In general, about 10% of the total emitted mass from aircraft gas turbine engines is emitted during takeoff and landing, except for VOCs and CO; 30% of the mass of these two chemicals is emitted during takeoff and landing (Federal Aviation Administration 2005).

A variety of gas turbine emissions are products of incomplete combustion (Kugele, Jelinek and Gaffal 2005). If aviation fuel were purely composed of hydrogen and carbon atoms and were entirely burned in air in an ideal combustor, only nitrogen oxides, water, and carbon dioxide would result. Because aviation fuel contains other compounds such as sulfur and is not burned entirely, other chemicals are emitted.

1.1.1. Nitrogen oxides (NO_x)

Nitrogen oxides are created during the high-temperature combustion of fuels in the presence of air, which contains nitrogen (Flagan and Seinfeld 1988). The class of nitrogen oxides consists of a variety of compounds that contain nitrogen atoms, such as nitric oxide (NO), nitrous oxide (N_2O), and nitrogen dioxide (NO_2) (U.S. Environmental Protection Agency 1993). Manufacturers of aircraft gas turbines must measure NO_x emissions from engines during testing for certification, as dictated by the International Civil Aviation Organization (ICAO) (International Civil Aviation Organization 2005). NO_2 is particularly important as a pollutant, as concentrations of NO_2 are regulated in the ambient air over the United States (Code of Federal Regulations 2007); see Section 1.4.1 of this thesis for further information on regulated ambient atmospheric chemicals.

1.1.2. Sulfur oxides (SO_x)

Sulfur oxides are created when fuels containing sulfur are burned (Flagan and Seinfeld 1988). Of the sulfur oxides, sulfur dioxide (SO_2) is particularly important, and ambient concentrations of SO_2 are regulated in the United States (Code of Federal Regulations 2007). Jet engine manufacturers do not have to measure SO_x from engines during certification testing, but the maximum sulfur content of aviation fuels is set at 3000 parts per million (ppm) by international voluntary standards (ASTM International 2007). Fuels are discussed further in Section 2.5 of this thesis.

1.1.3. Carbon monoxide (CO) and unburned hydrocarbons (HCs)

Carbon monoxide is created when carbon-based fuels undergo incomplete combustion in engines (Flagan and Seinfeld 1988). ICAO regulations dictate that engine manufacturers must measure CO emissions during certification testing (International Civil Aviation

Organization 2005). Unburned hydrocarbons like formaldehyde and benzene are created by incomplete combustion of hydrocarbon-based fuels (Kugele, Jelinek and Gaffal 2005, Yelvington, Herndon, Wormhoudt, et al. 2007), and engine manufacturers must measure HC content in engine exhaust for certification.

The class of hydrocarbons known as volatile organic compounds are hydrocarbon vapors emitted from certain solids or liquids (U.S. Environmental Protection Agency 1994a). VOCs are also emitted by biogenic sources, such as plants (Kesselmeier and Staudt 1999). Some VOCs cause severe health problems and are classified as hazardous air pollutants, as mentioned above.

1.1.4. Particulate matter (PM)

Particulate matter is composed of a combination of chemical components (species) of different sizes and compositions and is classified as primary PM or secondary PM (U.S. Environmental Protection Agency 2004a). Primary PM is created directly by combustion, mechanical abrasion, or erosion processes (Kugele, Jelinek and Gaffal 2005). Engine manufacturers do not have to measure primary PM directly when they test gas turbine engines for certification, but they must measure a quantity called smoke number (SN), which is related to visibility decreases caused by primary PM (International Civil Aviation Organization 2005).

Secondary PM is formed from chemical reactions involving NO_x, SO_x (particularly SO_2), VOCs, ammonia, and other compounds in the ambient atmosphere some time after these gases (known in this context as precursor emissions) are emitted by various sources (Kugele, Jelinek and Gaffal 2005, U.S. Environmental Protection Agency 2004a). For example, SO_2 and NO_x contribute to the formation of ammonium nitrate and ammonium sulfate particles in the atmosphere (U.S. Environmental Protection Agency 2004a). Engine manufacturers are not required to measure how emissions from their engines affect secondary particulate matter concentrations, but the relationships of these emissions to secondary ambient particulate matter concentrations may be estimated using air quality simulations.

Particulate matter is also classified as volatile or nonvolatile. Nonvolatile particles are particles made of elemental carbon or other materials like dust or metals; they are not chemically reactive. Volatile particles are created by gaseous precursors, like sulfuric acid, which can condense directly in the ambient air or may condense around condensation nuclei such as nonvolatile particles already in the atmosphere. The class of volatile PM includes semi-volatile particles, which exist in "an equilibrium between gaseous and condensed phases" (U.S. Environmental Protection Agency 2004a).

To measure concentrations of PM in the ambient air of the United States, the U.S. EPA uses PM monitors built to a Federal Reference Method (FRM) standard. This standard is described in (Code of Federal Regulations 1987) for PM of 10 μm in diameter and smaller (known as PM_{10}) and in (Code of Federal Regulations 2006) for PM of 2.5 μm in diameter and smaller ($PM_{2.5}$). The Federal Reference Method standard dictates the shape and other characteristics of PM monitors as well as appropriate techniques for measurement and handling. The EPA has also deployed numerous $PM_{2.5}$ species monitors around the United States in a network known as the Speciation Trends Network (STN) and also in Interagency Monitoring of Protected Visual Environments (IMPROVE) sites to determine trends in $PM_{2.5}$ species; the typical species measured by the EPA in these monitors are elemental carbon, organic carbon, ionic species (such as ammonium, nitrate, and sulfate), and heavy elements like metals. The monitors also collect water attached to particles. This particle-bound water plays an important role in particle size, thus influencing the nature of particle deposition in measuring equipment, on the surfaces of buildings and other objects, and inside of the human lung (U.S. Environmental Protection Agency 2004a).

Through its measurement networks, the EPA has determined that fine particles (below 2.5 μm) in the eastern and central United States are composed mostly of sulfate and organic carbon compounds; in the western U.S., $PM_{2.5}$ is composed of organic compounds and sulfate or nitrate (U.S. Environmental Protection Agency 2004b). Total

organic compounds constitute 10% to 70% of fine PM, but only 10% to 20% of these organic compounds can currently be quantified due to their chemical nature.

PM from aircraft activity consists of primary PM as well as secondary PM resulting from SO_x and NO_x emissions (Rojo 2007). Nonvolatile primary PM from aviation ranges from 0.02 to 0.06 μm in size with a lognormal distribution; volatile primary particles range from 0.001 to 0.015 μm in size with a lognormal distribution and are 10 to 100 times more numerous than nonvolatile PM (Lukachko, Waitz, Miake-Lye and Brown 2005). Thus, aviation PM is classified as $PM_{2.5}$. Only nonvolatile primary particulate matter can be found directly at a gas turbine engine's exit nozzle (Kugele, Jelinek and Gaffal 2005), but volatile primary particulate matter becomes important 30 meters behind the engine nozzle due to the conversion of fuel sulfur and unburned hydrocarbons into PM as well as the condensation of hydrocarbons and sulfur onto the surfaces of nonvolatile particles. Volatile primary PM can be a factor of 5 to 20 greater in number than nonvolatile particles at 30 meters of distance from the engine exit (Wey, Anderson, Wey, Miake-Lye, Whitefield and Howard 2007).

The uniqueness of aviation PM in comparison to other sources (such as automobiles) is still being determined. Aviation primary PM is unique in at least one specific way: it is deposited directly into the upper atmosphere by aircraft traveling at high altitudes (Wey, Anderson, Wey, Miake-Lye, Whitefield and Howard 2007). PM precursor emissions like SO_2 and NO_x from aircraft will react in the earth's atmospheric boundary layer (ABL, or mixing layer) in the same way as SO_2 and NO_x from other sources, so other unique characteristics of aviation PM depend on the uniqueness of the primary PM component. Primary PM is a minority of PM-related aviation emissions by mass; for instance, the U.S. Federal Aviation Administration's (FAA) System for Assessing Aviation's Global Emissions (SAGE) year 2005 data showed emissions of 600 tons of primary $PM_{2.5}$, 2,700 tons of SO_2, and 56,000 tons of NO_x due to aircraft flights from 2571 airports in the continental United States (Rojo 2007).

One way to discuss the uniqueness of aviation PM is by comparing the size distribution of aviation primary PM to the size distributions of PM from other sources. A study by (Harris and Maricq 2001) investigated a wide range of automobile engines and indicated that the total (nonvolatile and volatile) primary PM from the studied diesel engines was approximately of a lognormal size distribution with a mean ranging from 0.06 to 0.12 μm based on particle number; the studied gasoline engines had more asymmetric size distributions with means ranging from 0.04 to 0.08 μm. These measurements can be compared to measurements of nonvolatile primary PM from aviation (0.02 to 0.06 μm size with a lognormal distribution) and volatile primary PM from aviation (0.001 to 0.015 μm in size with a lognormal distribution).

1.1.5. Tropospheric Ozone (O_3)

Aircraft do not emit ozone into the earth's troposphere; tropospheric O_3 is formed entirely from precursor emissions (U.S. Environmental Protection Agency 2006b). Specifically, NO_x and VOCs react with sunlight to cause the formation or destruction of ozone in the atmosphere. CO also affects the ozone formation process in some cases. Though aviation-related CO emissions are of the same order of magnitude as aviation-related NO_x emissions, other sources produce much larger relative amounts of CO in comparison to aviation, making NO_x the most important aviation-related contributor to the ozone process (Penner 1999). In general, the ozone process has a strong nonlinear dependency upon concentrations of chemical precursors and their reaction rates as well as the intensity and spectral distribution of sunlight.

1.2. Emissions from other aviation-related sources

There are other aviation-related sources of emissions that must be considered when assessing the impact of aviation activity on air quality. The non-aircraft aviation-related sources usually considered are aircraft auxiliary power units (APUs), aircraft ground support equipment (GSEs), and ground access vehicles (GAVs). Stationary power sources like power plants at airports, the handling and storage of fuel (which can emit fuel-related vapors), and the activity of emergency response teams and training fires as

well as airport-specific construction are also aviation-related emission sources (Ratliff 2007).

Auxiliary power units are used to provide power for heating, air conditioning, and electrical systems when aircraft are on the ground. APUs embedded in aircraft are often small gas turbine engines. The usage of APUs depends on aircraft size, ground traffic, and weather conditions as well as procedures that are specific to each airline, and ultimate decisions on usage rest with an aircraft's pilot (Ohsfeldt, Waitz, Sequeira, et al. 2007). GSEs are also critical to airport operations; these are vehicles that assist with aircraft servicing procedures such as refueling and the loading and unloading of baggage. They can be powered by a range of sources, from diesel engines to electric motors (Morrow, Hochard and Francfort 2007). Ground access vehicles are road vehicles that pass through airport boundaries while transporting people and cargo; just like other vehicles currently on roadways, GAVs have a diverse set of power sources.

The apportionment of aviation-related emissions between aircraft and other sources is not generally known and has only been determined for specific airports. Heathrow airport in the United Kingdom is one airport where such an apportionment has been done. In the year 2002, airside vehicles (vehicles within the airport boundary, such as ground support equipment) contributed 5.2% of Heathrow's total NO_x emissions and 20.4% of the total PM_{10} emissions; stationary sources contributed another 3.9% of NO_x and 25.2% of PM_{10} in the same year (AEA Energy & Environment 2007). Emissions from car parking and car rental as well as taxi waiting areas added approximately 0.6% of NO_x and 1.8% of PM_{10}; emissions of vehicles entering and leaving the airport boundary were not assessed by the study.

1.3. The health impacts of aviation activity

1.3.1. Noise

Noise is one of the most immediately recognizable environmental effects of aviation. Aviation-related noise comes from aircraft themselves as well as airport-related ground

traffic and industrial noise, and noise levels in neighborhoods near airports may exceed 60 dB(A) (A-weighted decibels, a measure of sound pressure levels as experienced by humans) (Health Council of the Netherlands 1999). Studies have gathered evidence that exposure to aviation-related noise can cause hypertension, sleep disturbance, and degradations in the performance of children at school (Health Council of the Netherlands 1999).

1.3.2. Climate change

CO_2 emissions from human activity contribute to changes in the earth's climate (Forster, Ramaswamy, Artaxo, et al. 2007). Global warming is "likely to affect the health statuses of millions of people" due to increases in the frequency of floods, heat waves, storms, and fires. These changes, as well as changes in crop productivity and disease vector patterns, will increase health burdens worldwide (Intergovernmental Panel on Climate Change 2007). Aircraft can cause long-term perturbations in the climate through CO_2 emissions as well as short-term perturbations through PM, water, and NO_x emissions; in sum, aviation may be responsible for several percent of anthropogenic global warming (Penner 1999).

Aircraft emissions of water vapor can form condensation trails (contrails) in atmospheric conditions suitable for water vapor condensation, such as the low temperature conditions found at high altitudes (U.S. Environmental Protection Agency 2000a). Contrails that persist and become clouds after an aircraft has passed can also contribute to global warming by trapping heat radiated from the earth's surface. The formation of persistent contrails can be enhanced by volatile and nonvolatile primary PM from aircraft, as the particles can act as cloud condensation nuclei. Because NO_x is an ozone precursor, aircraft NO_x emissions affect ozone concentrations in the atmosphere. Aviation-induced changes in ozone concentrations in the middle tropospheric to lower stratospheric layers of the atmosphere may contribute to warming of the global climate. NO_x emissions also reduce the lifetime of atmospheric methane, leading to a cooling influence globally. The net balance between the warming and cooling influence of aviation NO_x emissions varies regionally and is still the subject of much scientific study.

1.3.3. Effects on skin cancer

Aviation emissions can also indirectly affect rates of human skin cancer through their influence on ozone levels at high altitudes, which varies the amount of UV (ultraviolet) radiation reaching earth from the sun. This changes skin cancer incidences (Brunelle-Yeung 2007). The relationship between UV and cutaneous malignant melanoma (CMM) skin cancer is "not completely understood," but researchers have a better understanding of the relationship between UV-B rays and non-melanoma skin cancer (NMSC). Subsonic aviation activity in the year 1992 may have reduced yearly NMSC-related deaths by 9 to 16 incidences.

1.3.4. Air quality

Aviation emissions below the atmospheric mixing height (including aviation-related emissions at ground level) are distributed throughout the earth's atmospheric boundary layer, changing the chemical composition of ambient air and coming in contact with people (Ratliff 2007). Researchers who study the effects of atmospheric pollutants on human health usually report their findings using the metrics of relative risk (RR), as in Equation (1), or odds ratio, as in Equation (2). In conjunction with a baseline incidence rate of a certain health effect (for example, all recorded cases of asthma in a certain population in a particular year), the increase or decrease in statistical health incidences due to a certain pollutant can be derived (Rojo 2007).

$$RR = \frac{y_0}{y_c} \tag{1}$$

$$\text{Odds Ratio} = RR \cdot \left(\frac{1 - y_c}{1 - y_0} \right) \tag{2}$$

y_0 and y_c are statistical health incidences observed at ambient concentrations C_0 and C_c.

Health effects of NO_x

NO_x is created from the combustion of fuels at high temperature in the presence of ambient air, which contains nitrogen. Mean concentrations of NO_2 were approximately 15 parts per billion (ppb) in Metropolitan Statistical Areas (MSAs) of the United States from 2003 to 2005; the highest maximum hourly concentrations measured were

approximately 200 ppb (U.S. Environmental Protection Agency 2007d). Direct exposure of humans to NO_2 has been demonstrated to cause effects in human lung defense systems at concentrations below 1000 ppb. For instance, a 2003 study by Chauhan et al. as cited by (U.S. Environmental Protection Agency 2007d) showed that "increased personal exposure to NO_2 worsens virus-associated symptoms and lung function in children with asthma" at low concentrations, "with medians for the exposure quartiles ranging from 2.6 to 10.9 ppb," though "confounding with [$PM_{2.5}$] emissions remains a concern." In general, there is a correlation between asthma symptoms in adults and children and short-term (minutes to hours) exposure to NO_2 at near-ambient levels; correlations between short-term exposure to near-ambient NO_2 levels and incidences of cardiovascular disease are currently less clear.

An EPA assessment of various studies investigating premature mortality due to short-term exposure to near-ambient NO_2 levels indicated that the "range of NO_2 total mortality risk estimates is 0.5 to 3.6% per 20-ppb increase in the 24-h average NO_2 (or 30-ppb increase in the daily 1-h maximum)" concentrations (U.S. Environmental Protection Agency 2007d). As a comparison, a Harvard study of six cities as cited by (U.S. Environmental Protection Agency 2004b) found a 4.2% to 23% increase in total premature mortality incidences per 10 $\mu g/m^3$ in ambient $PM_{2.5}$ concentrations. Assuming a density of 1.2 kg/m^3 for ambient air, this is approximately a 10% to 55% increase in premature mortality incidences per 20 ppb increase in ambient $PM_{2.5}$ concentrations. Health effects studies have indicated associations between chronic NO_2 exposure and respiratory health impacts, especially in children; the relationships between chronic NO_2 exposure and cardiovascular disease are less clear. There is currently a limited understanding of associations between chronic NO_2 exposure and premature mortality; this is due to a lack of data as well as confounding between the effects of NO_2 and other pollutants (such as particulate matter) in numerous studies.

A few studies have assessed the health effects of nitrous acid (HONO) and nitric acid (HNO_3). For instance, some associations have been observed between HONO concentrations and wheezing in asthmatic adults. For HNO_3, "very few toxicological

studies" have been done, but the studies that have been done suggest that nitric acid has similar toxicological effects to NO_2 (U.S. Environmental Protection Agency 2007d).

NO_x has an indirect health impact as a precursor to ozone and secondary particulate matter formation (Rojo 2007). This is important in the context of aviation-related emissions, as aircraft emit substantial amounts of NO_x. These emissions can contribute to concentrations of nitric acid in the atmosphere, and nitric acid reacts with ammonia in the ambient air to create small particles of ammonium nitrate (U.S. Environmental Protection Agency 2004a).

Health effects of SO_x

SO_x is created from the burning of fuels containing sulfur molecules. The health impacts of long-term direct exposure to SO_2 in ambient air are uncertain (U.S. Environmental Protection Agency 1986). Studies of peak exposure to SO_2 have focused primarily on respiratory symptoms in asthmatics, and affected asthmatics represent a small portion of the United States population (U.S. Environmental Protection Agency 1994b).

The indirect health impacts of SO_x (as a precursor to particulate matter) are stronger and more certain than the direct health impacts, as SO_x emissions contribute to the formation of ammonium sulfate particles. Increases in SO_x emissions can also reduce ambient concentrations of ammonium nitrate particles, as ammonia in the ambient air will preferentially react with sulfate (U.S. Environmental Protection Agency 2004a). This inhibition of ammonium nitrate formation due to SO_x emissions is sometimes called the "bounceback effect" (Rojo 2007).

Health effects of CO

CO is a product of the incomplete combustion of carbon-based fuels. Negative impacts of CO on the human cardiovascular system have been observed, especially in individuals with impaired cardiovascular systems (U.S. Environmental Protection Agency 2002). Many adverse health effects of CO did not occur in health studies until the CO concentration was much higher than typical ambient CO concentrations. The effects of

long-term human exposure to carbon monoxide are uncertain. CO is of some importance as an aviation-related emission because it can affect the creation of ozone, particularly in urban areas (U.S. Environmental Protection Agency 2006b).

Health effects of VOCs

VOCs are hydrocarbon vapors emitted from certain solids or liquids as well as biogenic sources. In the year 2001, 72% of the outdoor VOCs in the United States came from biogenic sources, as determined in the EPA's 2001 National Emissions Inventory (NEI) (U.S. Environmental Protection Agency 2005a) through usage of the Biogenic Emissions Inventory System (BEIS) modeling tool (U.S. Environmental Protection Agency, National Oceanic and Atmospheric Administration 2007). Due to VOC production by household products such as paints, varnishes, and furniture, indoor VOC concentrations are usually much higher than outdoor concentrations even in regions with significant outdoor air pollution sources (U.S. Environmental Protection Agency 1994a).

Many of the health effects of VOCs, such as nausea, fatigue, and throat irritation, are temporary (U.S. Environmental Protection Agency 1994a). However, there are VOCs that cause cancer or other serious health effects. These VOCs are classified as HAPs; chromium, formaldehyde, and benzene pose particularly severe nationwide carcinogenic risks (U.S. Environmental Protection Agency 2002). VOCs are important in the context of aviation emissions because they contribute to ozone formation by reacting with NO_x.

Health effects of PM

PM is a mixture of components of various sizes, sources, and compositions. The size of PM is important in the context of its health effects, and ambient PM exhibits several distinct size distributions, as shown in Figure 1 (U.S. Environmental Protection Agency 2004b). Many human health studies have focused on the health effects of thoracic PM, which is PM small enough to penetrate into the thoracic region of the human lower respiratory tract. PM_{10} (which includes $PM_{2.5}$) is considered to be thoracic PM, and coarse particles ($PM_{10\text{-}2.5}$) and fine particles ($PM_{2.5}$ and smaller) are often studied separately due to differences in formation, transformation, and removal processes as well

as differences in chemistry. Coarse and fine particles generally have higher fractional depositions (which is the fraction of inhaled particles that deposit in the human lung) than accumulation mode particles.

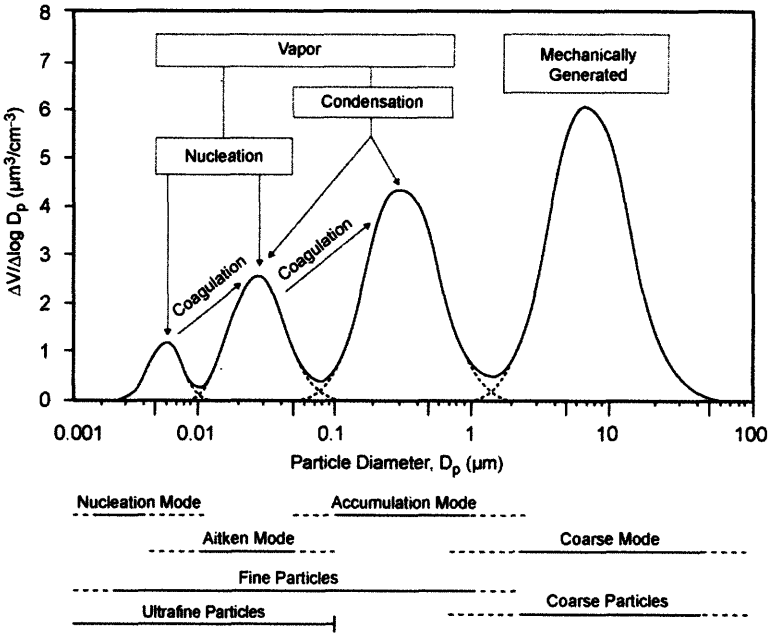

Figure 1: Size distributions of particulate matter (U.S. Environmental Protection Agency 2004a)

Increases in ambient concentrations of thoracic PM are strongly associated with increases in incidences of cardiovascular and respiratory diseases. Furthermore, numerous studies have indicated that long-term exposure to $PM_{2.5}$ is more significantly associated with mortality effects than short-term exposure to larger PM.

Cross-sectional studies and cohort studies have provided a good picture of the health effects of long-term exposure to PM. For example, a Harvard study of six cities found a strong association between long-term $PM_{2.5}$ exposure and mortality, as cited by (U.S. Environmental Protection Agency 2004b), and the researchers computed a total mortality excess RR of 4.2% to 23% for every 10 $\mu g/m^3$ increase in ambient $PM_{2.5}$ concentrations.

The effects of specific PM species on human mortality are uncertain; these effects have varied across cities and studies. Air quality modeling studies such as (Greco, Wilson,

Spengler and Levy 2007), (Rojo 2007), and (Levy, Wilson, Evans and Spengler 2003) have assumed that each species of particle has the same health damage per unit mass when estimating the health implications of different methods of modeling PM production and dispersion.

Health effects of ozone

Ozone is created by chemical reactions involving NO_x, VOCs, and sunlight; CO can also affect ozone formation. Laboratory experiments have linked ozone exposure with detrimental respiratory effects in animals and humans, including structural changes in certain regions of the lung, at concentrations similar to ambient ozone levels (U.S. Environmental Protection Agency 2006b). Several multi-city studies in the United States and Europe have shown "small, but very precise (extremely narrow 95% [confidence intervals]) positive associations for increased mortality risk using all-year ambient O_3 data and warm-season only data" as cited by (U.S. Environmental Protection Agency 2006b). Studies by Bell (2004) and Schwartz (2005) have also shown positive correlations between ambient ozone concentrations and mortality, as cited by (U.S. Environmental Protection Agency 2006b).

1.3.5. Monetary valuation of changes in health effects

For economic analyses of policies, changes in health effects can be monetized. The monetary valuation of changes in health effects related to air quality has been performed for a number of EPA rulemakings, such as the 2007 proposed revisions to the ozone National Ambient Air Quality Standards (NAAQS) (U.S. Environmental Protection Agency 2007a). A particularly common economic measure of health effects is a subpopulation's willingness to pay (WTP) for a reduction in risk of a certain health effect. WTP is the "maximum amount of money an individual would voluntarily exchange to obtain an improvement (or avoid a decrement) in the environmental effects of concern" (U.S. Environmental Protection Agency 2000d). The total economic benefits of a reduction in health effects are the sum of all affected individuals' willingness to pay for the reduction in that health effect.

The WTP metric cannot be used for all health effects; instead, the cost of illness (COI) is sometimes estimated. This metric generally addresses the direct cost of treatment for an illness. The costs in the form of pain and suffering are left un-quantified and thus not included. The benefits of a health-related policy are measured in the reduction in COI due to a change in a certain health effect.

The Value of Statistical Life (VSL) is also often used in policy analyses. VSL is an aggregation of many individuals' willingness to pay for small decreases in mortality risk. VSL is computed by taking an individual's WTP for a reduction in mortality and dividing that value by the change in mortality risk from the implementation of a policy. The EPA assessed a variety of VSL literature and computed the VSL in the United States as $5.5 million in 1999 dollars (with a 95% confidence interval of $1 million to $10 million) in its 2006 PM NAAQS Regulatory Impact Analysis (U.S. Environmental Protection Agency 2006a).

To compare benefits or costs that occur at different points in time, monetary discounting is often applied to benefits or costs that occur in the future. The implication of discounting is that "a given amount of future consumption is worth less than the same amount of consumption today" (U.S. Environmental Protection Agency 2000d). The EPA has applied 3% and 7% discount rates to its valuations of premature mortality and non-fatal myocardial infarction (heart attack) due to air pollutant exposure, as those health effects are assumed to occur years after exposure.

1.4. Health impacts of aviation in the U.S.: The Energy Policy Act of 2005

Section 753 of the Energy Policy Act of 2005 (United States Statutes at Large 2005) required "the Administrator of the Federal Aviation Administration and the Administrator of the Environmental Protection Agency [to] jointly initiate a study to identify:"

- "the impact of aircraft emissions on air quality in non-attainment areas:" non-attainment areas are described in reference to the National Ambient Air Quality Standards (NAAQS) set by the EPA in the United States.
- "ways to promote fuel conservation measures for aviation to enhance fuel efficiency and reduce emissions; and"
- "opportunities to reduce air traffic inefficiencies that increase fuel burn and emissions"

The study mandated by the Energy Policy Act explored aviation's effects on human health via changes in air quality. Several government and contracting organizations were involved, including the Massachusetts Institute of Technology. This thesis builds on the Energy Policy Act study by exploring how health impacts determined in the study are apportioned among various PM species. The thesis also assesses the assumptions used in the creation of the emissions inventory utilized by the study. It then analyzes how health impacts are related to aviation emissions regulations. The Energy Policy Act study and its results are described in detail in (Ratliff, Sequeira, Waitz, et al. 2008).

1.4.1. National Ambient Air Quality Standards in the United States

In the United States, the Clean Air Act requires the administrator of the U.S. EPA to publish "ambient air quality standards the attainment and maintenance of which . . . are requisite to protect the public health" (United States Code 2005g). These standards are known as the National Ambient Air Quality Standards (NAAQS). The NAAQS consist of primary and secondary standards; primary standards protect public health, while secondary standards protect public welfare issues such as visibility (United States Code 2005g). Air pollutants regulated by the NAAQS are known as criteria pollutants. A non-attainment area is "any area that does not meet (or that contributes to ambient air quality in a nearby area that does not meet) the national primary or secondary ambient air quality standard" for a criteria pollutant (United States Code 2005a). A maintenance area is an area that has left non-attainment for a particular pollutant and is implementing a maintenance plan to remain out of non-attainment status (United States Code 2005f).

The current criteria pollutants are defined in (Code of Federal Regulations 2007) and are listed in Table 1. Details on measurement methods for these pollutants are described in the same set of regulations.

Table 1: Current EPA-designated criteria pollutants and their standards of maximum concentration

Pollutant	Category	Statute	Standard
SO_2	Primary	40 CFR 50.4	0.030 ppm annual arithmetic mean; 0.14 ppm 24-hour average
SO_2	Secondary	40 CFR 50.5	0.5 ppm 3-hour average
PM_{10}	Primary	40 CFR 50.6	150 $\mu g/m^3$ 24-hour average
PM_{10}	Secondary	40 CFR 50.6	150 $\mu g/m^3$ 24-hour average
$PM_{2.5}$	Primary	40 CFR 50.13	15.0 $\mu g/m^3$ annual arithmetic mean; 35 $\mu g/m^3$ 24-hour average
$PM_{2.5}$	Secondary	40 CFR 50.13	15.0 $\mu g/m^3$ annual arithmetic mean; 35 $\mu g/m^3$ 24-hour average
CO	Primary	40 CFR 50.8	9 ppm 8-hour average; 35 ppm 1-hour average
Ozone[1]	Primary	40 CFR 50.9-50.10	0.12 ppm 1-hour average; 0.08 ppm daily maximum 8-hour average
Ozone[2]	Secondary	40 CFR 50.9-50.10	0.12 ppm 1-hour average; 0.08 ppm daily maximum 8-hour average
NO_2	Primary	40 CFR 50.11	0.053 ppm annual arithmetic mean
NO_2	Secondary	40 CFR 50.11	0.053 ppm annual arithmetic mean
Lead	Primary	40 CFR 50.12	1.5 $\mu g/m^3$ maximum arithmetic mean averaged over a calendar quarter
Lead	Secondary	40 CFR 50.12	1.5 $\mu g/m^3$ maximum arithmetic mean averaged over a calendar quarter

1.4.2. Data used in the Energy Policy Act Study

Numerous data sources were used in the Energy Policy Act study. These data sources provided aircraft emissions, population data, ambient concentrations of pollutants, and other information. The EPA 2001 National Emissions Inventory was used to obtain baseline emissions of CO, NO_x, VOCs, SO_2, NH_3 (ammonia), PM_{10}, and $PM_{2.5}$ (U.S. Environmental Protection Agency 2007f). Note that the 2001 NEI is an internal EPA

[1] The Environmental Protection Agency can designate that an area meet either the 1-hour or 8-hour ozone standards.
[2] See Footnote 1.

inventory based on the publicly released EPA 1999 NEI (Dolwick and Manning 2008). Monthly commercial aircraft emissions from 325 airports in the continental United States from June 2005 to May 2006 were estimated using the Emissions and Dispersion Modeling System (EDMS). Specifically, the emissions estimated were CO, VOCs, SO_2, NO_x, total primary $PM_{2.5}$, and individual sulfate, organic carbon, and elemental carbon $PM_{2.5}$ species. The EDMS emissions "capture 95 percent of nationwide activity of aircraft with engines certified to the International Civil Aviation Organization (ICAO) emission standards (specifically, those with ICAO smoke numbers)" (U.S. Environmental Protection Agency 2007f). Seven aircraft operating modes were modeled using EDMS: engine startup, auxiliary power units, aircraft taxiing to gates, aircraft taxiing out to runways, takeoff with initial climb, climb out, and approach. Note that EDMS generated emissions only below 3000 feet, an assumption for the thickness of the earth's atmospheric boundary layer.

Several inputs were provided to EDMS. Aircraft operations data came from the United States Bureau of Transportation Statistics (BTS) (Bureau of Transportation Statistics 2008), the Air Traffic Activity Data System (ATADS) (Federal Aviation Administration 2008), and FAA's Enhanced Traffic Management System (ETMS) (Federal Aviation Administration 2007a). Aircraft main engine and APU specifications, aircraft weight, and ground operating times were obtained from the BTS, the BACK Associates fleet database (BACK Aviation Solutions 2007), and the National Airspace System Resources (NASR) database (Federal Aviation Administration 2007b). A summary of inputs to EDMS is shown in Figure 2 (Ratliff, Sequeira, Waitz, et al. 2008). Additionally, the results of a survey of the APU usage for several airlines were used.

For the health impacts analysis in the Energy Policy Act study, a variety of health-related data was used. A dataset of the 2001 continental United States population from the year 2000 United States Census was used to determine human exposure to air pollutants (Dolwick 2007). Baseline health incidence and prevalence rates from the year 2000 were employed; these came from a variety of sources such as the Centers for Disease Control and Prevention as well as National Hospital Discharge Surveys (Davidson 2007, Ratliff,

Sequeira, Waitz, et al. 2008). The concentration-response functions for PM and ozone were provided by the EPA (Davidson 2006a, Davidson 2006b) and constructed using a variety of sources.

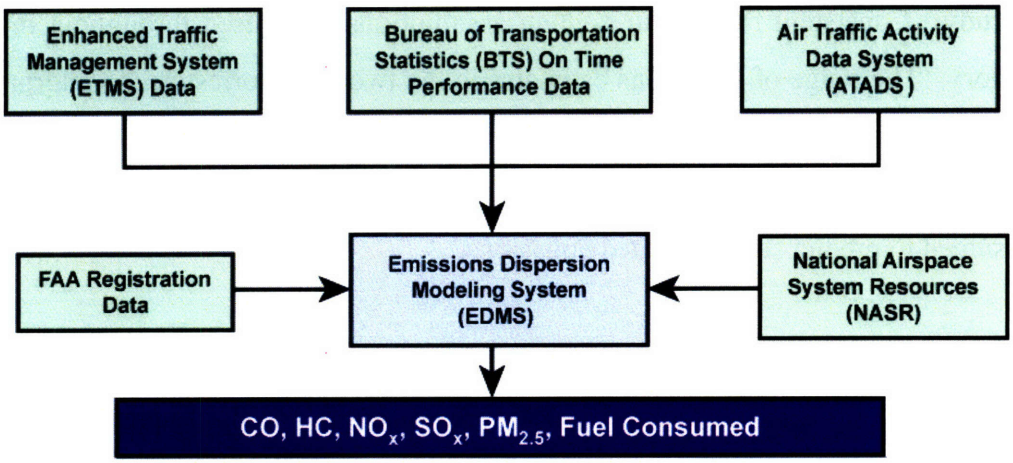

Figure 2: Summary of inputs to EDMS (Ratliff, Sequeira, Waitz, et al. 2008)

Finally, the study used U.S. meteorological data from 16 December 2000 to 21 March 2002 from the Pennsylvania State University / National Center for Atmospheric Research Mesoscale Model (MM5) version 3.6.1 (McNally 2003, U.S. Environmental Protection Agency 2007f). The meteorological data was used to model the transport, dispersion, and chemical transformation of aviation emissions.

1.4.3. Errors and conservatively biased assumptions in the Energy Policy Act study inventory

Because engine manufacturers do not measure primary PM from engines during certification (International Civil Aviation Organization 2005), the amount of primary PM from aircraft must be modeled. The primary PM modeling method used for the Energy Policy Act study is a modification of ICAO's First Order Approximation (FOA) version 3.0; this modification is known as FOA3a and is described along with FOA3 in (Ratliff, Sequeira, Waitz, et al. 2008). Lower fuel sulfur assumptions were used in approximately one quarter of the airports in the Energy Policy Act study, reducing SO_x emissions in the inventory by approximately 15%.

The Energy Policy Act inventory can be compared with an inventory built by CSSI (CSSI Inc. 2006), known as the Response Surface Model (RSM) inventory; the RSM inventory was created using FOA3 and the same aircraft operations data used in the Energy Policy Act study. Consistent fuel sulfur content assumptions were used throughout the RSM inventory. The range of emissions estimates in the two inventories can be interpreted as representing the general uncertainty in the estimation of aviation-related emissions, as scientific knowledge of primary PM is still being advanced and fuel sulfur content must be assumed to predict sulfate-related emissions.

FOA3a has a number of differences in comparison with FOA3. First, FOA3 uses an engine's certified smoke number (SN) at different operating modes to predict the mass of nonvolatile primary particulate matter produced per kilogram of fuel consumed (known as an Emission Index or EI, often stated in mg/kg fuel). FOA3a also uses SN to predict nonvolatile particulate matter; FOA3a, however, increases the SN by 3 and uses the bypass ratio (BPR) of an aircraft's engine as a conservative multiplier. This multiplier increases the EI by up to a factor of 8 (Ratliff, Sequeira, Waitz, et al. 2008). The BPR multiplier is applied to all engines in the modeled aircraft fleet; however, it is physically correct only for mixed-flow turbofans, which comprise a minority of the fleet (Waitz 2007).

FOA3 assumes that the mass of volatile primary PM per kilogram of fuel burned is a function of the fuel sulfur content, the content of fuel organic components, and the engine lubrication oil. A certain fraction of the fuel sulfur becomes sulfuric acid, which begins to condense to form volatile particulate matter by the time the exhaust plume has reached 30 meters of distance from the engine's exit nozzle. The rest of the fuel sulfur becomes SO_2. In FOA3a, an "upper limit of 5%" is used as the fraction of fuel sulfur that becomes sulfuric acid (Ratliff, Sequeira, Waitz, et al. 2008). By comparison, (Sorokin, Katragkou, Arnold, Busen and Schumann 2004) found that 1.3% to 3.3% of fuel sulfur may become sulfuric acid in an exhaust plume 5 ms in age.

Emissions of organics-related volatile primary PM scale directly with an engine's emissions of unburned hydrocarbons in FOA3. The FOA3 methodology acknowledges that engine lubrication oil may be in volatile primary particulate matter but does not include an estimate of this component, as lubrication oil contributions to volatile PM are not well documented. FOA3a implements conservative modifications for volatile primary PM from organics based on the results of the Aircraft Particle Emissions Experiment (APEX) (Wey, Anderson, Wey, Miake-Lye, Whitefield and Howard 2007) and assumes that 1.4 grams of lubrication oil is released per landing-takeoff (LTO) cycle. A more complete description of FOA3a and FOA3 is in (Ratliff, Sequeira, Waitz, et al. 2008).

After the analysis was done in the Energy Policy Act study, an error was discovered in the generation of the sulfur-related emissions in the Energy Policy Act inventory. Flights belonging to 78 airports (out of 325) were assigned a fuel sulfur content of 400 ppm instead of 680 ppm, which is more typical of average jet fuel sulfur content, causing a reduction in the inventories of SO_x and sulfate-related volatile primary PM (Melissa Ohsfeldt, Ian Waitz 2007).

The Energy Policy Act inventory's conservatively biased assumptions and error in the sulfate-related emissions can be compared to the RSM inventory produced by CSSI, Inc. In this inventory, the BPR multiplier was applied only to mixed-flow turbofans in the simulated fleet. SO_x emissions at all airports were computed using an assumed fuel sulfur content of 680 ppm. A comparison of nonvolatile, organics-related volatile, and sulfate-related volatile primary PM masses in the two inventories is shown in Figure 3.

In comparison with the RSM inventory, the nonvolatile primary PM mass in the Energy Policy Act inventory was higher by a factor of 3.4, while the organics-related and sulfate-related volatile primary PM masses were higher by a factor of 3.5 and 10.2, respectively. This led to a difference in total primary PM mass of a factor of 4.4.

A comparison of the difference in SO_x inventories is shown in Figure 4. In comparison with the RSM inventory, the EPAct inventory had 15% less SO_x by mass. A small and unexplained difference between the NOx inventories in the Energy Policy Act and the RSM was also identified; the Energy Policy Act inventory had a 0.2% higher mass of NO_x in comparison with the RSM. The differences in the inventories lead to differences in modeled health effects, discussed further in Section 1.4.8 of this thesis.

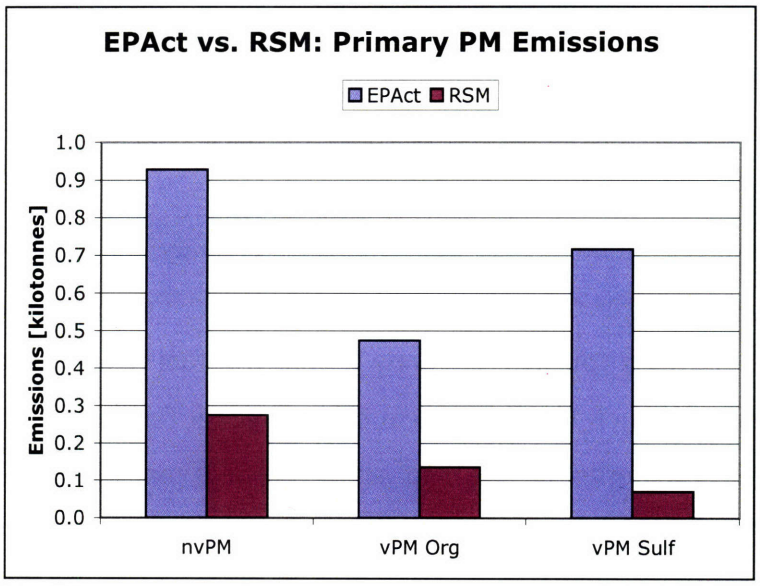

Figure 3: Comparison of primary PM masses in the Energy Policy Act and RSM inventories

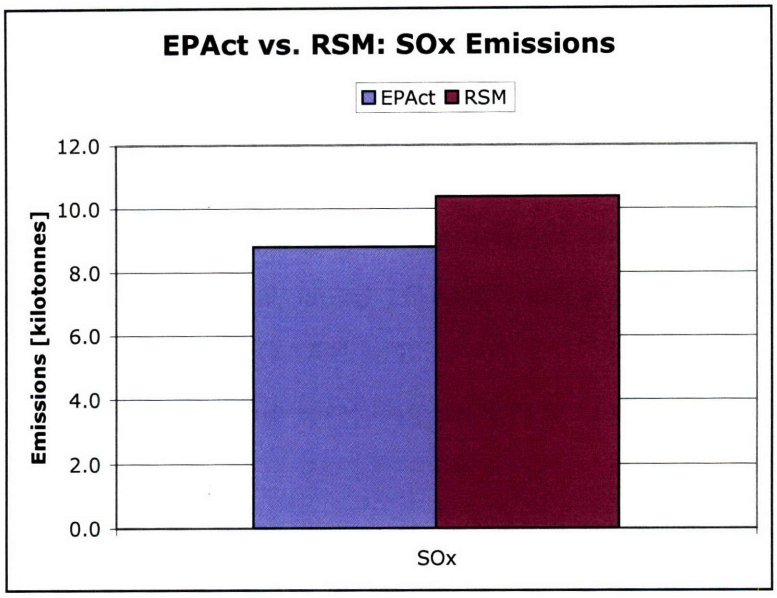

Figure 4: Comparison of the SO_x emissions mass in the Energy Policy Act and RSM inventories

1.4.4. Air quality analysis methodology in the Energy Policy Act study

Proper modeling of pollutant transport, dispersion, and transformation is important in determining human exposure and ultimate health effects. For the Energy Policy Act study, the behaviors of oxidant precursors and particulate matter concentrations over the continental United States were modeled using the Community Multiscale Air Quality Model (CMAQ). The CMAQ simulation employed a 36 km horizontal grid resolution and 14 vertical layers extending from the ground to an altitude representing 100 millibars of atmospheric pressure. The simulation used the 2001 EPA NEI baseline emissions, 2001 meteorological data from MM5, and lateral boundary conditions and initial species concentrations from the GEOS-CHEM atmospheric chemistry model. GEOS-CHEM was executed using a 2.0 x 2.5 degree (latitude-longitude) horizontal resolution and 20 vertical layers. Note that the 2001 NEI is an inventory that is internal to EPA and is based on the publicly available 1999 EPA NEI (Dolwick and Manning 2008).

Changes in PM and ozone concentrations were estimated by first creating a baseline scenario using the 2001 NEI. Then, the results of a simulation without aircraft emissions and a simulation with the EDMS inventory were used to scale concentrations computed

in the baseline scenario. The Speciated Model Attainment Test (SMAT) process was performed on the PM model output based on a 5-year (1999 – 2003) average of PM design values at Federal Reference Monitor (FRM) sites, where the middle years are weighted more heavily. A design value is a "statistic that describes the air quality status of a given area relative to the level of the National Ambient Air Quality Standards" (U.S. Environmental Protection Agency 2007f). Ozone was treated using an ozone-specific relative reduction factor (RRF) method. The NAAQS are described in Section 1.4.1 of this thesis, while the SMAT and RRF processes are described in (Ratliff, Sequeira, Waitz, et al. 2008).

The baseline scenario consisted of 2001 NEI emissions and included emissions listed in the six NEI source classification categories (SCCs) representing aviation:

- 2275000000: Aircraft: All Types and Operations
- 2275001000: Aircraft: Military Aircraft
- 2275020000: Aircraft: Commercial Aircraft
- 2275050000: Aircraft: General Aviation
- 2275060000: Aircraft: Air Taxi
- 2275070000: Aircraft: Auxiliary Power Units

The baseline scenario was used to obtain ambient PM and ozone design values. The next scenario was a no-aircraft scenario, where the NEI emissions from the aviation-related SCCs were removed. Finally, an EDMS aircraft scenario was created and used as input. In this scenario, NEI aviation emissions were removed and the year 2005 commercial aircraft emissions from EDMS were added. The CMAQ output of PM and ozone concentrations from the no-aircraft scenario and the EDMS aircraft scenario were used in a relative sense to adjust ambient design values obtained in the baseline scenario.

1.4.5. Health impact analysis methodology in the Energy Policy Act study

The health impact analysis used a methodology consistent with that employed for the PM and ozone NAAQS regulatory impact analyses (Ratliff, Sequeira, Waitz, et al. 2008). It utilized the Environmental Benefits Mapping and Analysis Program (BenMAP) from Abt Associates, Inc., version 2.4.85, built on 7 May 2007. Figure 5 contains a high-level visual overview of the BenMAP methodology and inputs (Abt Associates Inc. 2005). The pollutants considered were $PM_{2.5}$ and ozone.

The health impacts of PM and ozone were analyzed separately. For each pollutant, the difference in concentrations between the CMAQ no-aircraft and CMAQ EDMS aircraft scenarios was computed. To estimate the resulting change in human exposure, the year 2001 continental United States population dataset was input to BenMAP. BenMAP used the change in human exposure in conjunction with year 2000 baseline incidence and prevalence rates of a variety of illnesses to compute changes in human health incidences due to pollutant exposure.

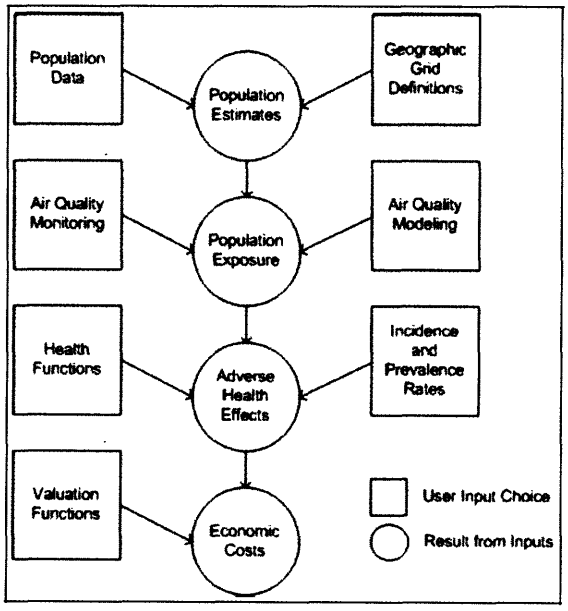

Figure 5: Summary of BenMAP inputs and methodology (Abt Associates Inc. 2006)

Monetary valuation was not done as a part of the Energy Policy Act study, but valuations were computed for this thesis using BenMAP based on configurations provided by the EPA (Davidson 2006a, Davidson 2006b). The valuations were computed in year 2001 dollars using EPA standard inflators and an income growth adjustment dataset for the year 2001 (the dataset itself was dated 21 March 2007). For further information, see Appendix A: Valuation of health effects.

1.4.6. Significant findings of the Energy Policy Act study

The impact of 2005 EDMS aircraft on the average 2001 baseline 8-hour ozone design values is shown in Table 2 and Figure 6. By comparison, EPA's estimated policy-relevant background concentration for ozone (which is the concentration that would exist if all anthropogenic emissions of ozone precursors were removed in the United States) for ozone varies from 15 to 35 ppb (U.S. Environmental Protection Agency 2007e). Negative values in the table and figure indicate ozone decreases due to the removal of aviation, and positive values indicate ozone increases (known as disbenefits) due to the removal of aviation. Ozone is created by complex chemical reactions and forms more readily when NO_x and VOCs are present in similar proportions. VOCs often come from biogenic sources (Kesselmeier and Staudt 1999), so VOC concentrations can be low in city centers. Reductions in NO_x emissions in such locations bring NO_x and VOCs closer to similar proportions, increasing ozone concentrations.

Table 2: Impact of EDMS aviation on average national 8-hour ozone design values (U.S. Environmental Protection Agency 2007f)

	With aircraft emissions (ppb)	Without aircraft emissions (ppb)	Percent Change
Non-Attainment Areas[3]	91.21	91.10	-0.12%
All Counties[4]	84.95	84.85	-0.12%

[3] 126 ozone non-attainment areas.
[4] 645 counties with base year ozone monitoring data.

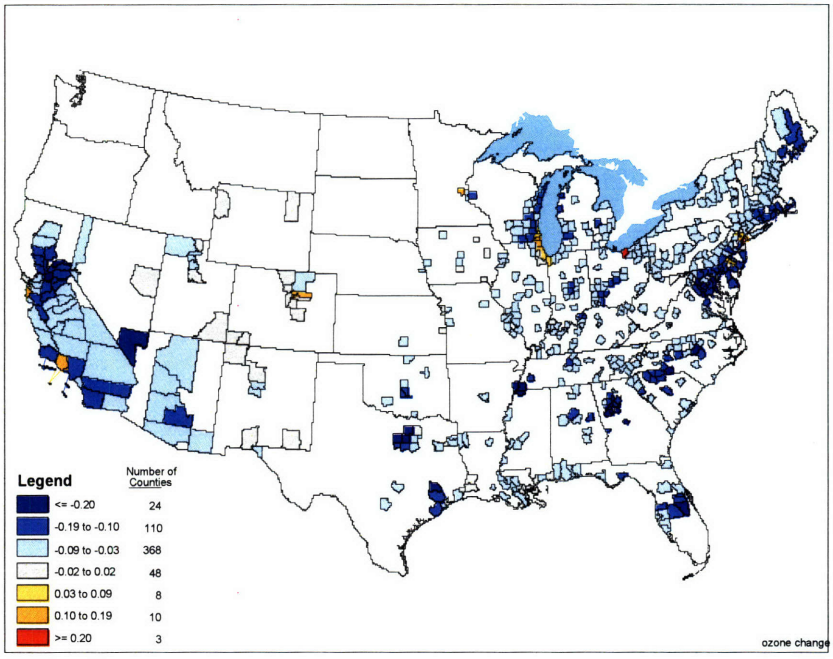

Figure 6: Change in 8-hour ozone concentrations (ppb) resulting from the removal of EDMS aircraft (U.S. Environmental Protection Agency 2007f)

The impact of 2005 EDMS aircraft on average 2001 baseline annual total $PM_{2.5}$ design values is shown in Table 3 and Figure 7. By comparison, EPA's estimated policy-relevant background for $PM_{2.5}$ is 3.5 $\mu g/m^3$ for urban areas in the eastern United States and 2.5 $\mu g/m^3$ for urban areas in the western United States (U.S. Environmental Protection Agency 2005c).

Table 3: Impact of aviation on average annual national total $PM_{2.5}$ design values (U.S. Environmental Protection Agency 2007f)

	With aircraft emissions ($\mu g/m^3$)	**Without aircraft emissions** ($\mu g/m^3$)	**Percent Change**
Non-Attainment Areas[5]	17.76	17.75	-0.06%
All Counties[6]	12.60	12.59	-0.08%

[5] 39 $PM_{2.5}$ non-attainment areas.
[6] 557 counties with base year $PM_{2.5}$ monitoring data.

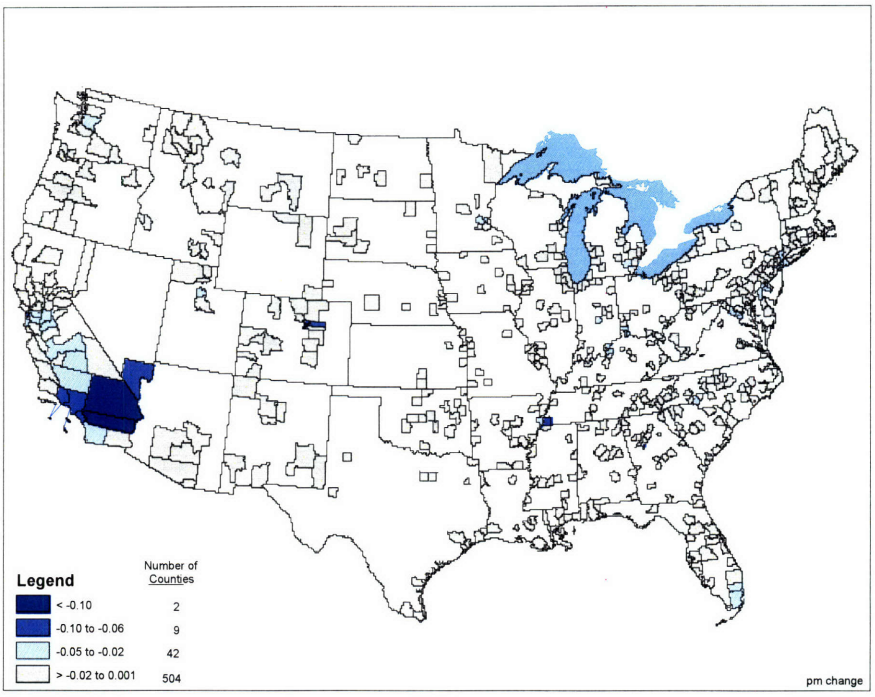

Figure 7: Change in annual total $PM_{2.5}$ concentrations ($\mu g/m^3$) resulting from the removal of EDMS aircraft inventories (U.S. Environmental Protection Agency 2007f)

1.4.7. The geographic distribution of impacts on air quality

Aviation's impacts on air quality were determined to have a strong regional nature. A histogram of aviation activity's contribution to the 2001 National Emissions Inventory for 273 counties in the continental United States (representing the 325 airports in non-attainment and maintenance areas[7]) is shown in Figure 8. Compared to the EDMS aviation scenario, the national average change in 8-hour ozone concentrations in the without-aviation scenario in the Energy Policy Act study's CMAQ simulation was a reduction of 0.10 ppb. The largest 8-hour ozone reduction occurred near the Atlanta, GA area, which saw a 0.31% decrease in ozone concentrations from 96.3 to 96.0 ppb.

[7] Emission from airports in the same FIPS area were aggregated. Data unavailable for nine FIPS areas.

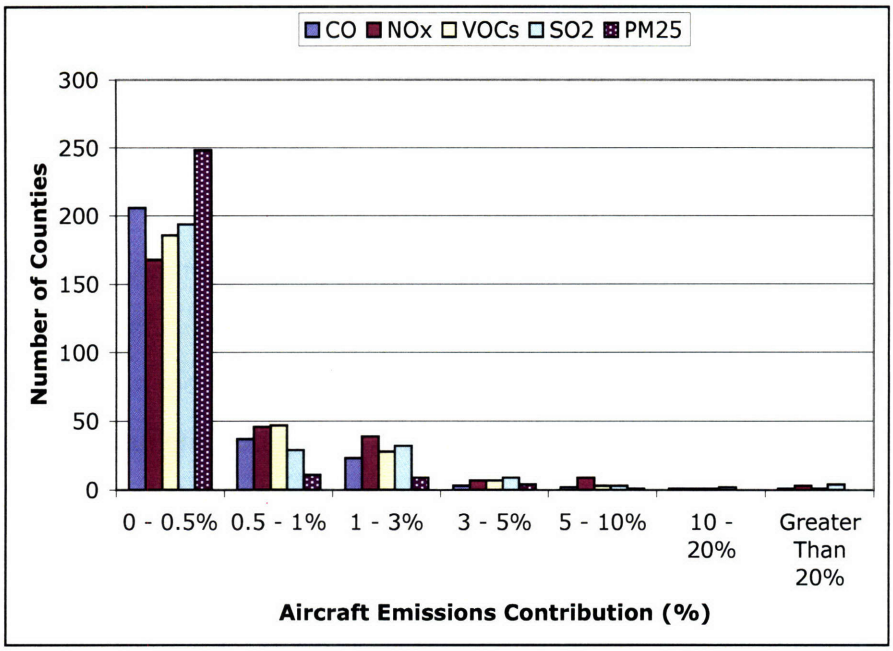

Figure 8: Percent contribution of commercial aircraft to the 2001 county-level emissions inventories for 273 counties (Ratliff, Sequeira, Waitz, et al. 2008)

In 24 counties in the continental United States, ozone increases (disbenefits) were observed in the without-aviation scenario. The largest increase occurred in Richmond County, NY, which experienced a 0.62% increase from 95.9 to 96.5 ppb in the CMAQ simulation. The national average change in $PM_{2.5}$ concentrations in the without-aviation scenario was a reduction of 0.01 $\mu g/m^3$. The largest reduction for $PM_{2.5}$ occurred in Riverside County, CA, which experienced a 0.52% decrease from 28.88 to 28.73 $\mu g/m^3$.

Several PM species compose the PM concentrations related to aircraft emissions. Ammonium nitrate and ammonium sulfate concentrations were estimated from the Energy Policy Act results using Equations (3) and (4) based upon apportionment equations received from the EPA (Timin 2007). These equations were based upon the SANDWICH technique described in (Frank 2006) and in EPA modeling guidance documentation (U.S. Environmental Protection Agency 2007c).

$$C_{AN} = 1.29 * 1.12 * C_{NO_3} \tag{3}$$

$$C_{AS} = C_{SO_4} + C_{NH_4} - 0.29 C_{NO_3} + C_{H_2O} - 0.12 * 1.29 * C_{NO_3} \tag{4}$$

C_{AN} is the concentration of ammonium nitrate, C_{NO_3} is the nitrate concentration, C_{AS} is the ammonium sulfate concentration, C_{SO_4} is the sulfate concentration, C_{NH_4} is the ammonium concentration, and C_{H_2O} is the particle-bound water concentration. Ammonium sulfate and ammonium nitrate molecules attract water, and the amount attracted has a strong nonlinear dependence upon ambient relative humidity (U.S. Environmental Protection Agency 2004a). While particle-bound water mass is not included when measuring total PM mass for determining non-attainment status, the water mass was used in the health impacts calculation for the Energy Policy Act study. This is because health studies reference FRM measurements, which include water mass (U.S. Environmental Protection Agency 2007c). The SANDWICH technique assigns 12% of the particle-bound water to ammonium nitrate.

The ambient concentrations of nonvolatile primary PM and organics-related volatile primary PM were then computed using Equations (5) and (6) as follows:

$$C_{nvPM} = C_{EC} + C_{Crustal} \tag{5}$$

$$C_{vPM_{Org}} = C_{Total} - \left(C_{nvPM} + C_{AN} + C_{AS}\right) \tag{6}$$

C_{nvPM} is the nonvolatile PM concentration, C_{EC} and $C_{Crustal}$ are concentrations of elemental carbon and crustal materials like metals (respectively), $C_{vPM_{Org}}$ is the organics-related volatile primary PM concentration, and C_{Total} is the concentration of total PM. Note that sulfate-related volatile primary PM emissions from aviation contribute to ambient concentrations of ammonium sulfate.

The species of PM that makes up the largest portion of contributions from aviation to ambient concentrations in each county is shown in Figure 9. In the majority of counties, ammonium sulfate dominates.

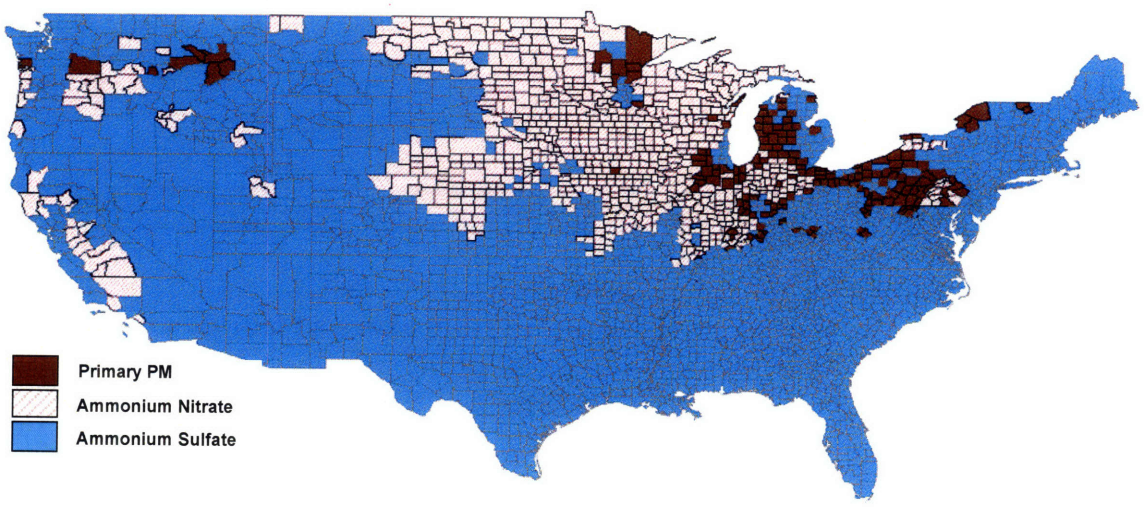

Figure 9: Largest aviation-related contributor to ambient PM concentrations in U.S. counties

1.4.8. Health effects of aviation in the Energy Policy Act study

The health impacts of aviation in the continental United States as computed by BenMAP are shown in Table 4; a list of concentration-response functions and valuation information can be found in (Ratliff, Sequeira, Waitz, et al. 2008) and in Appendix A: Valuation of health effects. A breakdown of health costs in the continental United States is shown in Figure 10.[8] Approximately 97% of the aviation-related health costs come from premature mortality of adults age 30 and over due to exposure to $PM_{2.5}$; premature mortality from ozone exposure is negative, and infant mortality due to $PM_{2.5}$ exposure is comparatively small, as shown in Table 4.

Table 4: Continental U.S.-level health impacts of commercial aviation in the continental United States (Ratliff, Sequeira, Waitz, et al. 2008)

[8] 3% discount rate used for premature mortality from PM for adults age 30 and over as well as for non-fatal myocardial infarction. Missing costs were assumed small (see Footnotes 16 - 21 and Footnotes 23, 25, and 26). From (Ratliff, Sequeira, Waitz, et al. 2008): Levy et al. 2005 meta-analysis study used for mortality from ozone.

Health Effect	Yearly Baseline Incidence	Yearly Cost (millions)[9] (91% Confidence Interval)	Yearly Incidences[10] (90% Confidence Interval)
PM-Related Endpoints:			
Premature mortality			
Adult, age 30 and over, 3% discount rate	2,300,000	$882 ($196 – $1830)	160 (64 – 270)
Adult, age 30 and over, 7% discount rate	2,300,000	$794 ($176 - $1650)	160 (64 – 270)
Infant, age <1	9,000	$3 ($1 – $5)	0 (0 – 1)
Chronic bronchitis (adult, age 27 – 99)	630,000	$40 ($3 – $139)	110 (20 – 200)
Non-fatal myocardial infarction (adult, age 18 - 99), 3% discount rate[11]	780,000	$26 ($7 – $57)	290 (160 – 430)
Non-fatal myocardial infarction (adult, age 18 - 99), 7% discount rate	780,000	$25 ($6 – $56)	290 (160 – 430)
Hospital admissions–respiratory (adult, age 0 – 64)[12]	640,000	$0 ($0 – $0)	26 (12 – 39)
Hospital admissions–respiratory (adult, age 65 – 99)[13]	570,000	$0 ($0 – $0)	12 (6 – 16)
Hospital admissions–cardiovascular (adult, age 18 – 64 for incidences, 20 – 64 for valuation)[14]	1,400,000	$1 ($0 - $1)	24 (14 – 34)
Hospital admissions–cardiovascular (adult, age 65 – 99)[15]	2,500,000	$1 ($1 - $1)	45 (29 – 60)

[9] 2001 U.S. dollars. Rounded to the nearest whole number and to three significant figures where applicable. 4.5th and 95.5th percentiles presented as given by BenMAP. Valuation done only for this thesis and not for Energy Policy Act study.

[10] Difference between CMAQ with EDMS aviation and without-aviation scenarios. Rounded to the nearest whole number and to two significant figures where applicable.

[11] Discounting applied as done for EPA's 2005 Clean Air Interstate Rule regulatory impact analysis (U.S. Environmental Protection Agency 2005b).

[12] From (Ratliff, Sequeira, Waitz, et al. 2008): Respiratory hospital admissions ages 0 – 64 for PM include admissions for chronic obstructive pulmonary disease (COPD) and asthma.

[13] From (Ratliff, Sequeira, Waitz, et al. 2008): Respiratory hospital admissions ages 65 – 99 for PM include admissions for COPD and pneumonia.

[14] From (Ratliff, Sequeira, Waitz, et al. 2008): Cardiovascular admissions include cardiovascular ailments except for myocardial infarctions.

Health Effect	Yearly Baseline Incidence	Yearly Cost (millions)[9] (91% Confidence Interval)	Yearly Incidences[10] (90% Confidence Interval)
Emergency room visits for asthma (age 0 - 17) [16]	730,000		140 (81 – 194)
Acute bronchitis (children, age 8-12)[17]	880,000		340 (-12 – 700)
Upper respiratory symptoms (asthmatic children, age 9 - 11)[18]	87,000,000		2,700 (860 – 4,600)
Lower respiratory symptoms (asthmatic children, age 7 - 14)[19]	14,000,000		3,700 (1,800 – 5,700)
Asthma exacerbation (asthmatic children, age 6 - 18)[20]	130,000,000		3,300 (370 – 9,600)
Work loss days (adults, age 18 - 64 for incidences, age 18 - 65 for valuation)	380,000,000	$3 ($3 – $3)	23,000 (20,000 – 25,000)
Minor restricted activity days (MRADs) (adults, age 18 - 64 for incidences)[21]	1,400,000,000		130,000 (110,000 – 150,000)
Ozone-Related Endpoints:			
Premature Mortality[22] (all ages) Bell et al. (2004)	930,000	-$2 (-$5 – -$1)	0 (0 – -1)

[15] From (Ratliff, Sequeira, Waitz, et al. 2008): Cardiovascular admissions include cardiovascular ailments and subcategories for ischemic heart disease, dysrhythmia and heart failure. Myocardial infarctions not included.

[16] Valuation unavailable for this age range; however, reductions in emergency room visits for respiratory ailments for individuals age 0 – 99 would save $0.04 million using a cost of illness method proposed by (Smith, Malone, Lawson, Okamoto, Battista and Saunders 1997).

[17] Valuation unavailable for this age range; however, reductions in cases of acute bronchitis for individuals age 0 – 17 would save approximately $0.13 million using an assumption of a 6-day illness.

[18] Valuation unavailable for this age range; however, reductions in upper respiratory symptoms for individuals age 0 – 17 would save approximately $0.07 million.

[19] Valuation unavailable for this age range; however, reductions in lower respiratory symptoms for individuals age 0 – 17 would save approximately $0.06 million.

[20] Valuation unavailable for this age range; however, reductions in asthma exacerbation for individuals age 0 – 17 would save approximately $0.15 million.

[21] Valuation unavailable for this endpoint; however, EPA's 2005 Clean Air Interstate Rule Regulatory Impact Analysis determined that a reduction of 8,100,000 MRADs (by reducing $PM_{2.5}$ and ozone) for adults age 18 – 65 saved $422 million.

[22] From (Ratliff, Sequeira, Waitz, et al. 2008): Consistent with the methodology used in the 2007 Ozone Regulatory Impact Analysis, ozone mortality estimates are included with the recognition that the exact magnitude of the effects estimate is subject to continuing uncertainty. Effect estimates from Bell et al. (2004) as well as effect estimates from three meta-analyses are given. An effect estimate of zero is also given to account for the possibility that there is no causal association between ozone and mortality.

	Health Effect	Yearly Baseline Incidence	Yearly Cost (millions)[9] (91% Confidence Interval)	Yearly Incidences[10] (90% Confidence Interval)
Meta-Analyses	Bell et al. (2005)	1,000,000	-$9 (-$18 – -$2)	-2 (-1 – -2)
	Levy et al. (2005)	1,000,000	-$11 (-$20 – -$3)	-2 (-1 – -2)
	Ito et al. (2005)	930,000	-$11 (-$19 – -$3)	-2 (-1 – -2)
Assume no causality between ozone and mortality		0	$0 ($0 – $0)	0 (0 – 0)
Hospital admissions–respiratory causes (adults, age 65 – 99)[23]		450,000		-3 (-5 – 0)
Hospital admissions–respiratory causes (children, age 0 – 1 for incidences, age 0 – 2 for valuation)[24]		180,000	-$0 (-$0 – -$0)	-6 (-3 – -10)
Emergency room visits for asthma (age 0 – 99)		710,000	-$ (-$0 – $0)	-4 (-12 – 0)
Minor restricted activity days (MRADs) (adults, age 18 – 65)[25]		570,000,000		-7,500 (-3,800 – -11,000)
School absence days (children, age 6 – 11)[26]		3,200,000,000		-2,800 (-4,700 – -990)

[23] Valuation unavailable for this age range. From (Ratliff, Sequeira, Waitz, et al. 2008): Respiratory hospital admissions for ozone include admissions for all respiratory causes and subcategories for COPD and pneumonia.

[24] From (Ratliff, Sequeira, Waitz, et al. 2008): Respiratory hospital admissions for acute respiratory diseases.

[25] Valuation unavailable for this endpoint.

[26] Valuation unavailable for this age range; however, the reduction in aviation would cause an increase in school absence days for individuals age 0 – 17 due to ozone disbenefits, leading to a cost of $0.2 million.

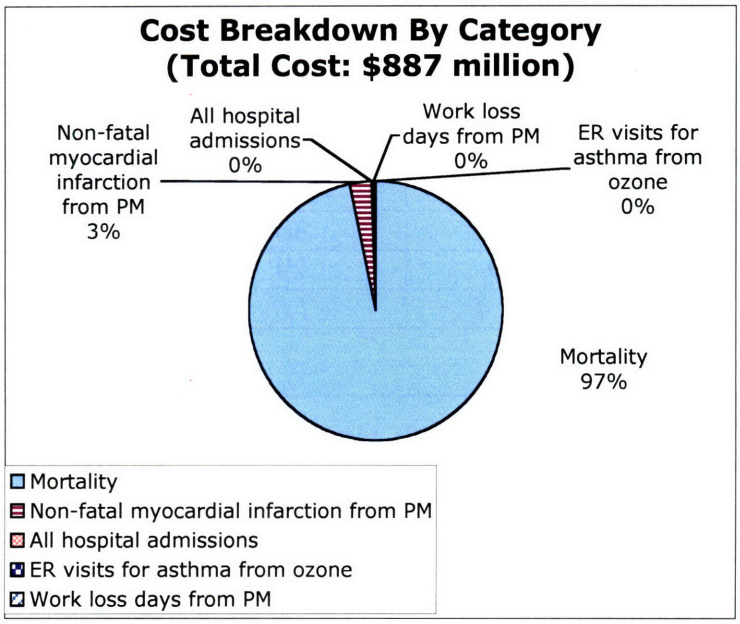

Figure 10: Breakdown of health costs from aviation-related emissions for the Energy Policy Act study

A model was developed to enable comparison of inventories as well as species apportionment; this model is described further in Section 3.3.1 of this thesis. Confidence intervals were not computed by the model.

The breakdown of incidences of premature mortality from PM by various aviation-related concentrations of primary and secondary PM is shown in Table 5 and Figure 11 for premature mortality of adults age 30 and over, while the breakdown of health costs is shown in Table 6 and Figure 12. The CRF used was (Pope, Burnett, Thun, et al. 2002). Health impact predictions based on the Energy Policy Act and RSM inventories are presented side-by-side. Total health impacts were estimated to be 140 incidences ($767 million) based on the RSM inventory. The relative apportionment of PM species concentrations to health incidences can be seen in Figure 13; the apportionment of the health costs is the same.

Table 5: Yearly mortality incidences based on the Energy Policy Act and RSM inventories[27]

Inventory	Nonvolatile Primary PM	vPM Organics	Ammonium Nitrate	SO_x Ammonium Sulfate	vPM Ammonium Sulfate
EPAct	23	30	30	75	6
RSM	8	9	29	97	1
Ratio	*2.9*	*3.4*	*1.0*	*0.8*	*10.3*

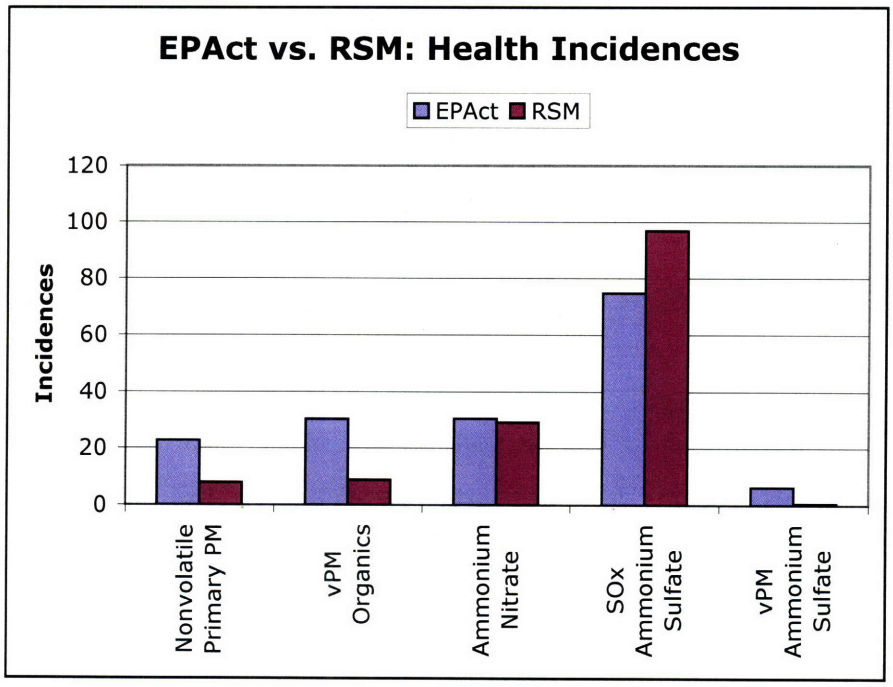

Figure 11: Apportionment of incidences of adult premature mortality among different $PM_{2.5}$ species for the Energy Policy Act and RSM inventories

Table 6: Yearly health costs based on the Energy Policy Act and RSM inventories, in millions[28]

Inventory	Nonvolatile Primary PM	vPM Organics	Ammonium Nitrate	SO_x Ammonium Sulfate	vPM Ammonium Sulfate
EPAct	$122	$162	$163	$401	$34
RSM	$42	$47	$156	$518	$3
Ratio	*2.9*	*3.4*	*1.0*	*0.8*	*10.3*

[27] Incidences rounded to the nearest whole number and to two significant figures where applicable.
[28] 2001 U.S. dollars. Costs rounded to the nearest whole number and to three significant figures where applicable.

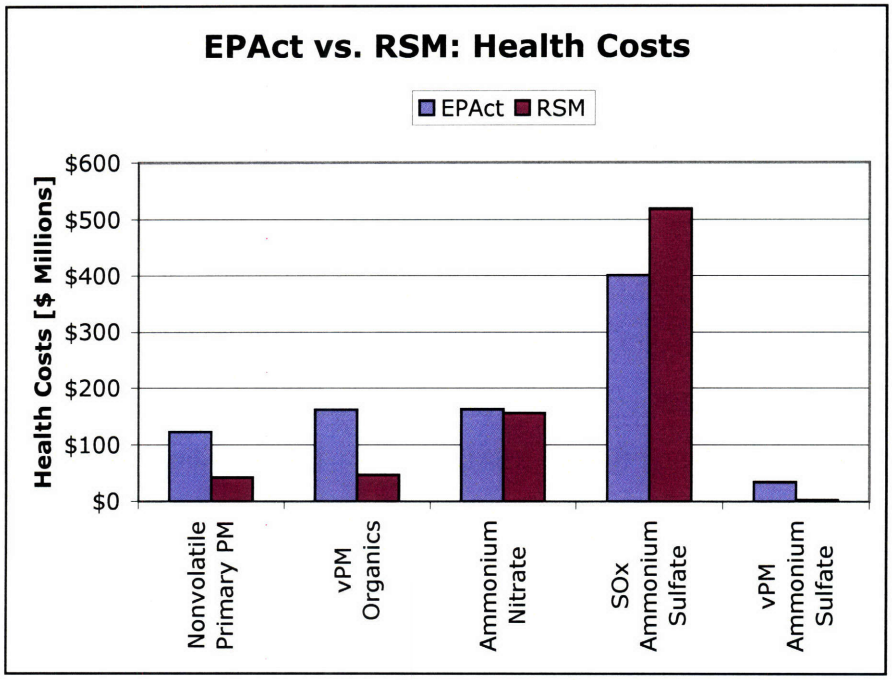

Figure 12: Apportionment of yearly costs of adult premature mortality among different $PM_{2.5}$ species for the Energy Policy Act and RSM inventories

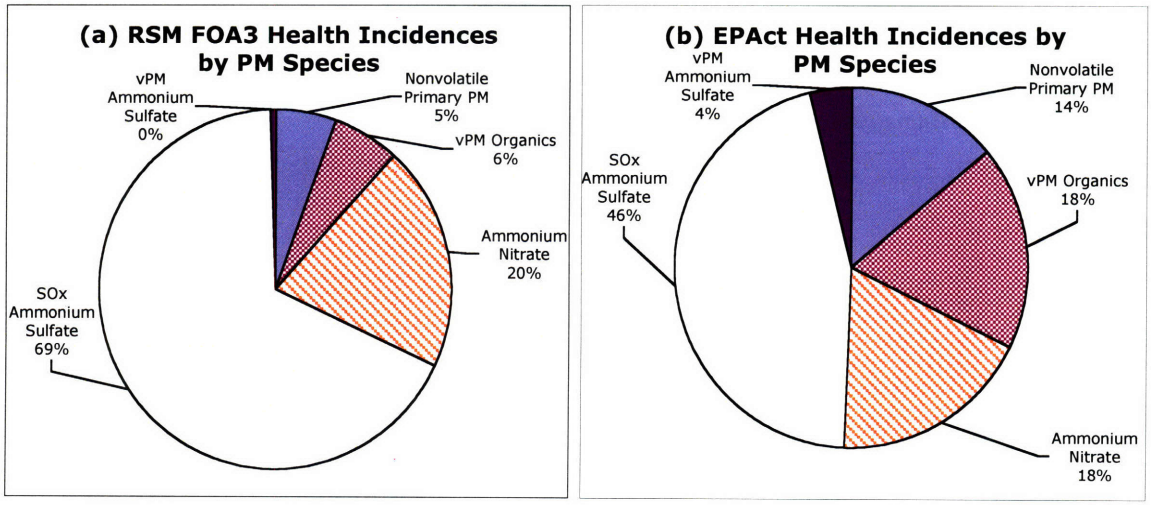

Figure 13: Relative apportionment of health incidences to various species of ambient PM concentrations

A more detailed breakdown of the health impacts by species is shown for the Energy Policy Act inventory in Table 7 and Figure 14, where the relative magnitudes of emissions as well as health effects can be compared. The marginal damage of each pollutant relates health costs from PM to unit emissions of primary PM or the precursors SO_x and NO_x. The bounceback-related reduction in health costs were allocated to SO_x marginal damages, as SO_x emissions increase health costs from ammonium sulfate but can simultaneously reduce health costs by reducing ammonium nitrate concentrations. The net cost per tonne of SO_x emitted can therefore be smaller than if the bounceback effect were neglected. Marginal damages are shown for the Energy Policy Act and RSM inventories in Figure 14 and for the RSM in Figure 15. The damages are similar across the two inventories.

Table 7: Marginal damages of emissions for the Energy Policy Act and RSM inventories in dollars per kilogram

Inventory	Nonvolatile Primary PM	vPM Organics	Ammonium Nitrate	SO_x Ammonium Sulfate	vPM Ammonium Sulfate
EPAct	$132	$340	$2	$43	$47
RSM	$152	$346	$2	$46	$47
Ratio	*0.9*	*1.0*	*1.0*	*0.9*	*1.0*

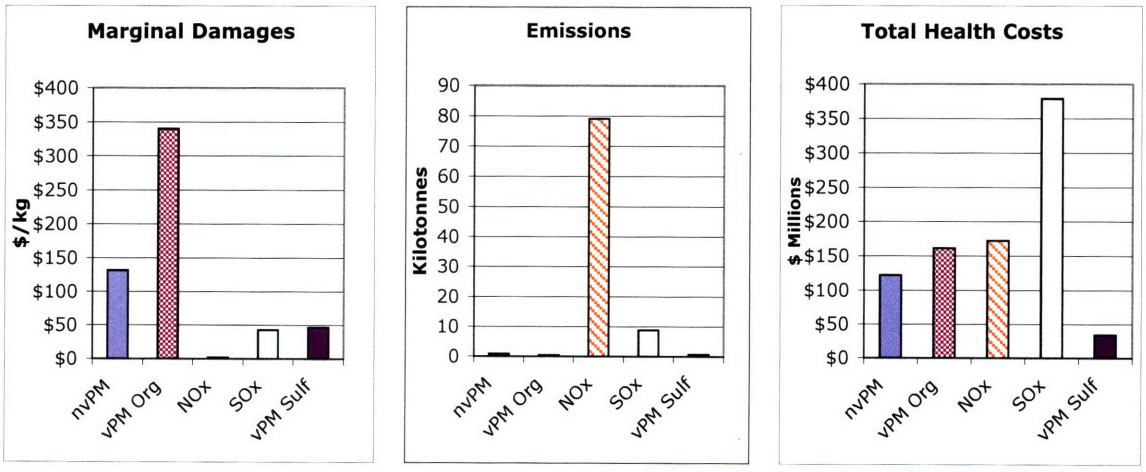

Figure 14: Marginal damages of aircraft PM-related emissions vs. the total health costs from various categories of aviation-related PM as found for the Energy Policy Act inventory

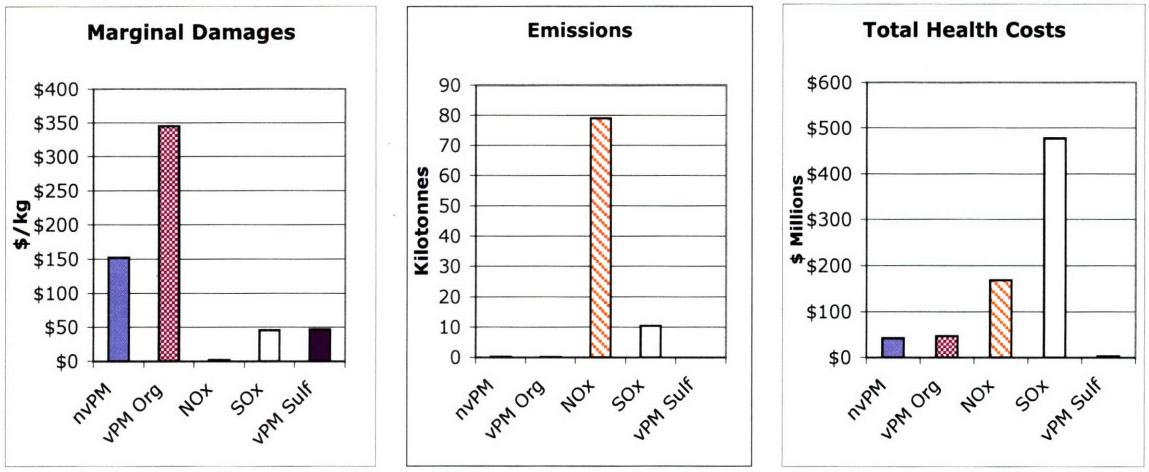

Figure 15: Marginal damages of aircraft PM-related emissions vs. the total health costs as found for the RSM inventory

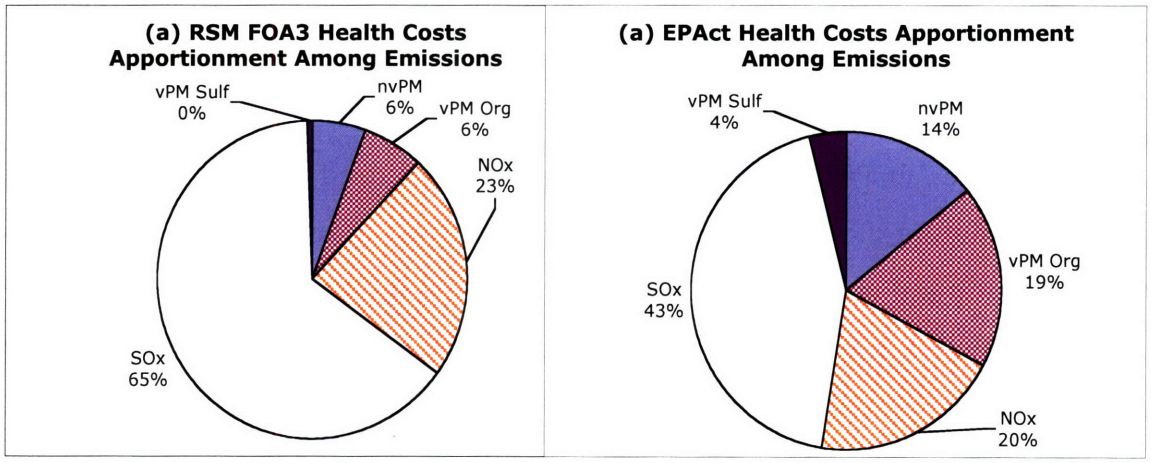

Figure 16: Health costs apportionment among the Energy Policy Act and RSM emissions inventories

The range of health incidence estimates implied by the conservatively biased Energy Policy Act inventory and the RSM inventory can be interpreted as representing uncertainty in the prediction of aviation-related emissions. Scientific knowledge of the nature of primary PM emissions and how to model it is still being advanced. The FOA3 and FOA3a methods implement less conservative and more conservative scientific assumptions concerning the creation of primary PM, respectively. Differences in the SO_x inventories can also be viewed as representing uncertainty in the creation of inventories

themselves, as fuel sulfur levels must be assumed in order to predict the masses of sulfate-related emissions due to aviation.

1.4.9. Geographic distribution of premature mortality incidences in the Energy Policy Act study

The ultimate geographic location of health impacts caused by emissions is a function of the location of the source and the physical transport and chemical transformation of those emissions; these latter processes depend on weather, temperature, and other factors (Greco, Wilson, Spengler and Levy 2007). The geographic distribution of health effects estimated in the Energy Policy Act study is presented below. Continental U.S.-wide health impacts from ozone are probably below the range of air quality model uncertainty (Dolwick, Davidson and Sequeira 2007), so a county-level apportionment would not give useful information for ozone and thus is not shown.

The distribution of PM health impacts across the ten United States counties with the greatest number of incidences is shown in Table 8 for premature mortality of adults age 30 and over, based upon the Pope et al. 2002 cohort study as used in the Energy Policy Act study (Ratliff, Sequeira, Waitz, et al. 2008). Los Angeles County was responsible for 18% of the PM mortality incidences and health costs due to aviation in the continental United States. The ranking based on the RSM inventory is shown in Table 9.

Table 8: Ten counties with highest PM mortality incidences in the Energy Policy Act study (Ratliff, Sequeira, Waitz, et al. 2008).

Rank	County	State	Incidences (Percent of Total)[29]	Cost in Millions (Percent of Total)[30]
1	Los Angeles	CA	29 (18%)	$155 (18%)
2	Orange	CA	8 (5%)	$43 (5%)

[29] Rounded to whole numbers and two significant figures where appropriate.
[30] Rounded to whole numbers and three significant figures where appropriate. Valuation only included for this thesis and not for Energy Policy Act study.

Rank	County	State	Incidences (Percent of Total)[29]	Cost in Millions (Percent of Total)[30]
3	San Diego	CA	6 (3%)	$29 (3%)
4	San Bernardino	CA	5 (3%)	$29 (3%)
5	Cook	IL	5 (3%)	$27 (3%)
6	Riverside	CA	4 (3%)	$23 (3%)
7	Nassau	NY	4 (3%)	$23 (3%)
8	Alameda	CA	4 (2%)	$20 (2%)
9	Queens	NY	3 (2%)	$16 (2%)
10	Kings	NY	3 (2%)	$15 (2%)
	All other counties		*94 (57%)*	*$502 (57%)*

Table 9: Ten counties with highest PM mortality incidences based on the RSM inventory

Rank	County	State	Incidences (Percent of Total)[31]	Cost in Millions (Percent of Total)[32]
1	Los Angeles	CA	30 (18%)	$160 (18%)
2	Orange	CA	9 (5%)	$47 (5%)
3	San Diego	CA	6 (4%)	$34 (4%)
4	San Bernardino	CA	6 (4%)	$32 (4%)
5	Riverside	CA	5 (3%)	$27 (3%)
6	Cook	IL	4 (3%)	$24 (3%)

[31] Rounded to whole numbers and two significant figures where appropriate.
[32] Rounded to whole numbers and three significant figures where appropriate. Valuation only included for this thesis and not for Energy Policy Act study.

Rank	County	State	Incidences (Percent of Total)[31]	Cost in Millions (Percent of Total)[32]
7	Alameda	CA	2 (3%)	$22 (3%)
8	Nassau	NY	3 (2%)	$17 (2%)
9	Queens	NY	2 (1%)	$12 (1%)
10	Kings	NY	2 (1%)	$11 (1%)
	All other counties		*71 (50%)*	*$381 (50%)*

The apportionment of health incidences to concentrations of various PM species for Los Angeles County is shown in Figure 17 for both inventories; the health costs apportionment is the same.

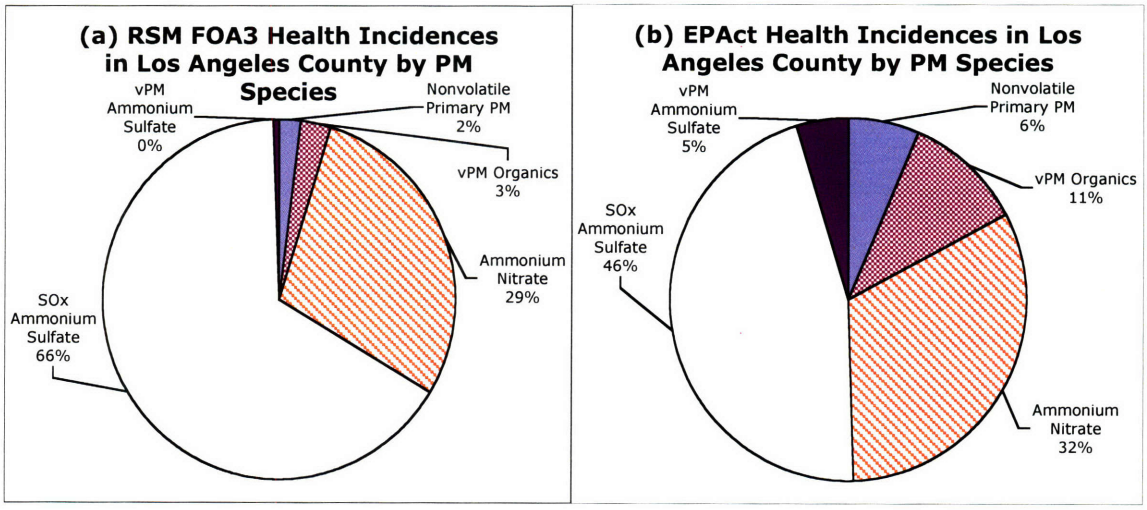

Figure 17: Apportionment of adult premature mortality health costs in Los Angeles County to concentrations of different PM species caused by aviation emissions, based on the Energy Policy Act and RSM inventories

1.4.10. Aviation's health impacts in context

An analysis of incidences of premature mortality caused by all anthropogenic $PM_{2.5}$ in the United States was not done for the Energy Policy Act study, but the yearly number of

incidences from anthropogenic PM is "very likely greater than 25000," meaning that premature mortality incidences due to aviation are 0.6% of this number or less (Ratliff, Sequeira, Waitz, et al. 2008). This number was estimated by comparing background United States $PM_{2.5}$ levels to levels of anthropogenic U.S. $PM_{2.5}$ and using an estimated premature mortality rate for adults age 30 and over of a 1% increase in mortality per $\mu g/m^3$ increase in exposure to ambient $PM_{2.5}$, based on expert elicitation studies (Levy and Waitz 2007) In comparison, the implementation of EPA's Tier 2, Heavy Duty Diesel, and Nonroad Diesel Engine rules are estimated to prevent a combined 25000 premature deaths from $PM_{2.5}$ by some time between 2020 and 2030 (U.S. Environmental Protection Agency 1999, U.S. Environmental Protection Agency 2000c, U.S. Environmental Protection Agency 2004d).

1.4.11. Uncertainty in measurement and apportionment of various particulate matter species

The interpretation of the results of the Energy Policy Act study and other studies using measured particulate matter depend on the accuracy of PM measurements. Unfortunately, there is no widely recognized PM measurement standard and also no widely recognized way to treat particle-bound water (U.S. Environmental Protection Agency 2004a). A variety of monitor types and techniques to measure PM are actively being developed and improved, and each has its strengths and weaknesses. Instead, the EPA defines "accuracy" based on how well a particular monitor agrees with a Federal Reference Method (FRM), which is updated and changed over time. The FRM is described in (Code of Federal Regulations 1987) for PM_{10} and in (Code of Federal Regulations 2006) for $PM_{2.5}$. FRM samplers for $PM_{2.5}$ must meet specific design and performance specifications and must also be handled in particular ways during measurement.

It is known that FRM techniques do not capture all ambient semi-volatile PM species, leading to a possible underestimate of total PM mass (U.S. Environmental Protection Agency 2007c). Semi-volatile materials (such as nitrates and organic compounds) have a

tendency to evaporate from PM monitoring devices when outside conditions change, such as when the monitor is moved for sampling (U.S. Environmental Protection Agency 2004a). In particular, nitrate is known to evaporate from Federal Reference Method monitors that use Teflon, and the negative effect of this on measurement accuracy is greater for $PM_{2.5}$ than it is for PM_{10}. The amount of mass lost has varied between studies but could be approximately 10% to 20%; for this reason, masses of PM species are reconstructed using the SANDWICH method (Frank 2006). In some cases, however, a substantial amount of nitrate can be lost during handling of the monitor itself, leading to an underestimate of ammonium nitrate concentrations.

It has also been determined that semi-volatile organic compounds can be lost in monitors, though there is currently little information to determine the magnitude of this loss. This is particularly important because organic compounds are one of the largest contributors to total PM mass. Additionally, it is possible to have positive artifacts in measurements of $PM_{2.5}$ due to the adsorption of gases onto filters inside of PM monitors. Using multiple types of filters (such as quartz and Teflon) can help to quantify these positive artifacts; Kim et al. 2001 estimated positive artifacts of organic carbon to be 30% annual average this way, as cited by (U.S. Environmental Protection Agency 2004a).

Furthermore, "a clear classification scheme has not yet been established to distinguish [organic carbon], light-absorbing carbon, black carbon, soot and [elemental carbon]" (U.S. Environmental Protection Agency 2004a). This causes difficulties in interpreting PM speciation data from monitors. The common classification in usage is as follows:

- "Organic carbon" category: semi-volatile organic carbon and non-visible light-absorbing carbon
- "Elemental carbon" category: elemental carbon, black carbon, soot, and light-absorbing carbon

The amount of particle-bound water (PBW) in $PM_{2.5}$ measurements is also hard to predict and control. While PBW mass is not included in the mass of PM that is subject to the

NAAQS, water affects the dispersion and deposition of particles; thus, it has implications for human health. Particle-bound water is a nonlinear function of relative humidity and other parameters and also exhibits hysteresis, as it condenses and vaporizes at different relative humidities. Removing particle-bound water by heating also leads to the loss of semi-volatile PM components. For these reasons, the mass of particle-bound water must be estimated. It is important to note that water mass is not measured directly; water is estimated using an inorganic aerosol model that is applied to the laboratory conditions of 21^{O} Celsius and 35% relative humidity in which FRM filters are measured. Because epidemiology studies reference EPA's FRM measurements, which include estimated particle-bound water mass, the water mass was included in the inputs to BenMAP for the Energy Policy Act study and also for the inputs to the apportionment model.

The SANDWICH method described in (Frank 2006) is a way of reconstructing the amounts of ammonium nitrate, ammonium sulfate, carbonaceous mass, and other PM components (such as metals) and is designed to compensate for some of the known biases of PM filters. This method is recommended by EPA in its model guidance document (U.S. Environmental Protection Agency 2007c) and was used for PM species apportionment in this thesis. The reconstructed ammonium nitrate and ammonium sulfate mass includes particle-bound water, which is predicted using an inorganic aerosol model. The amount of water is higher when fewer nitrates are present or when the collected aerosol is more acidic.

The minimum mass ratio is 12% water, which occurs in cases of nitrate-dominated aerosols; for this reason, the SANDWICH method apportions 12% of the water to ammonium nitrate. To explore the effects of particle-bound water apportionment, the apportionment model used in this thesis was executed again using an assumption that 50% of the water mass goes to nitrates. The effects of this assumption on health incidences apportionment are shown in Figure 18 for the continental United States and in Figure 20 for Los Angeles County; the apportionment of health costs is the same. The results using a 12% assumption are shown in Figure 19 and Figure 21. As can be seen, ammonium sulfate still remained as the dominant contributor to health incidences across

the nation. In Los Angeles County, ammonium nitrate concentrations slightly dominate over ammonium sulfate concentrations. 50% is an upper bound assumption, as the amount of particle-bound water is "much more dependent on sulfate concentration compared to nitrate" and there "is proportionally less estimated [particle-bound water] for wintertime aerosol which has higher NO_3 and lower SO_4" (U.S. Environmental Protection Agency 2007c).

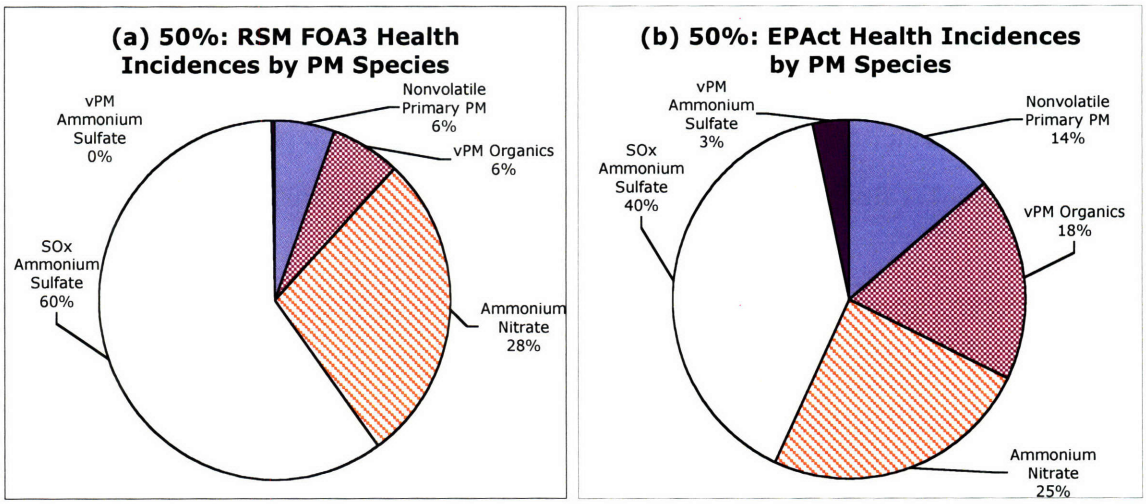

Figure 18: Apportionment of continental U.S.-wide health incidences when 50% of the water mass is assigned to nitrates, based on Energy Policy Act and RSM inventories

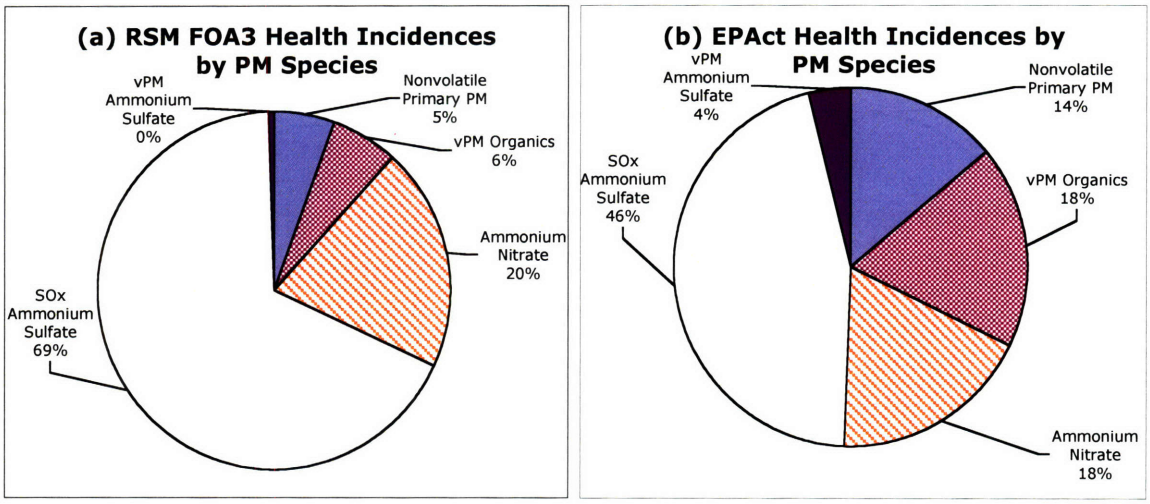

Figure 19: Review: Apportionment of continental U.S.-wide health incidences when 12% of the water mass is assigned to nitrates, based on Energy Policy Act and RSM inventories

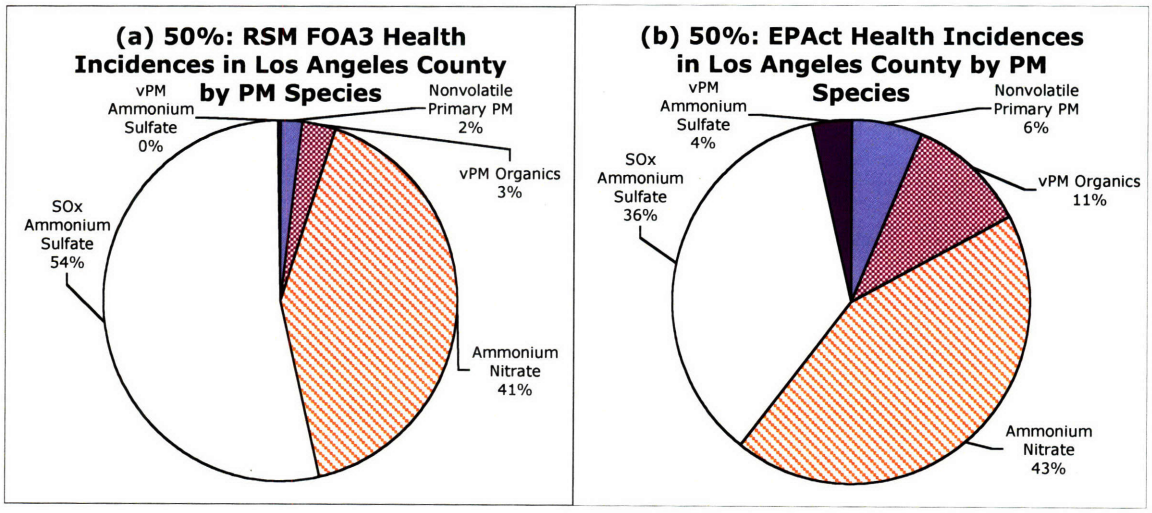

Figure 20: Apportionment of health incidences in Los Angeles County when 50% of the water mass is assigned to nitrates, based on Energy Policy Act and RSM inventories

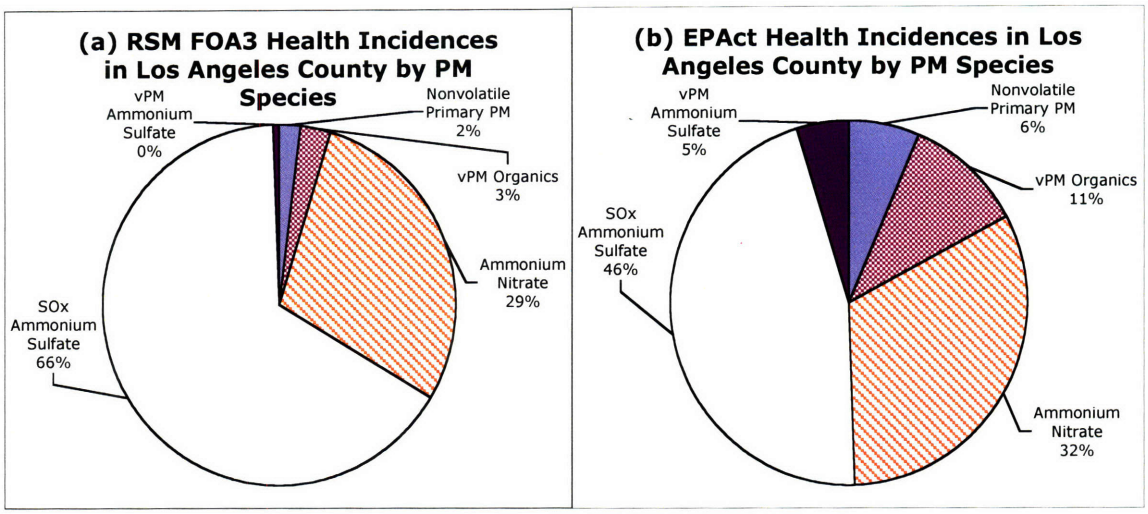

Figure 21: Review: Apportionment of health incidences in Los Angeles County when 12% of the water mass is assigned to nitrates, based on Energy Policy Act and RSM inventories

It is difficult to apportion organic carbon, so the SANDWICH method uses a mass balance technique to compute organic carbon mass. This technique essentially states that the organic carbon is the total PM mass minus the sum of the other species and minus the nominal "blank mass" value of 0.5. Blank mass is a category used for what appears to be filter contamination that is not correlated with the mass of PM in the ambient air (U.S. Environmental Protection Agency 2007c). Because the same nominal value is applied to all measurements, it essentially disappears when one estimates the contribution of a specific source (such as aviation) by subtracting out measured (or modeled) background concentrations of PM. The main issue of uncertainty is whether or not some or all of the PM mass counted as blank mass belongs in another category, as the nominal value of 0.5 is approximately the same magnitude as the measured nitrates, elemental carbon, and crustal matter in the Energy Policy Act study.

1.5. Review

This chapter presented and discussed the various compounds produced by aviation-related activity and their known health effects. It is known that aviation influences ambient concentrations of NO_x, SO_x, CO, HCs, tropospheric ozone, and PM. Aircraft emit NO_x, SO_x, CO, HCs, and some species of PM directly. SO_x influences

concentrations of secondary PM; NO_x and HCs influence secondary PM concentrations as well as ozone. Activities related to aviation, such as the operation of ground support equipment and ground access vehicles, also air quality; in general, their contribution to aviation-related emissions is not well known.

Exposure to ambient ozone and PM has been found to cause incidences of premature mortality. While the health effects of ambient concentrations of NO_x, SO_x, CO, and HCs are less certain, the roles of these compounds as precursor emissions for PM or ozone make them important to consider. The Energy Policy Act study considered the health effects of $PM_{2.5}$ and ozone and estimated that exposure to aviation-related ambient PM concentrations may cause 160 yearly incidences of premature mortality (90% CI of 64 to 270 incidences). The Energy Policy Act emissions inventory was built using FOA3a and had an error in the mass of SO_x; the effects of these were quantified in a comparison with the RSM inventory by CSSI, Inc., built using FOA3. Based on the comparison, it was estimated that aviation activity could be responsible for 140 to 160 incidences of premature mortality from exposure to aviation-related PM concentrations. 46% to 69% of the incidences was due to changes in concentrations of ammonium sulfate secondary PM from SO_x, while ammonium nitrate secondary PM was responsible for 18% to 20%. Exposure to volatile primary PM from organic compounds and nonvolatile primary PM were responsible for 6% to 18% and 5% - 14% of the impact, respectively, while volatile primary PM from sulfates was responsible for 0% to 4%. Confidence intervals were not computed, and only the effects of changes in PM concentrations were assessed. Strong regional effects were also observed; for instance, 18% of the total premature mortality incidences may have occurred in Los Angeles County.

Finally, the uncertainties and assumptions in EPA's current methods of monitoring PM were investigated. It is known, for example, that semi-volatile compounds such as nitrates will readily evaporate from FRM monitors. Total PM mass as well as the masses of various PM species must be reconstructed based on recorded values, and the chapter briefly discussed the employment of EPA's SANDWICH method to reconstruct ambient concentrations of PM. The chapter also discussed the SANDWICH method's

apportionment of 12% of particle-bound water to nitrates. Using an assumption of 50% particle-bound water to the nitrates, it was determined that changes in ammonium sulfate concentrations due to aviation still dominated the health impacts across the continental United States. However, ammonium nitrate slightly dominated over ammonium sulfate in Los Angeles County. The next chapter will review current regulations governing aviation emissions in order to give background for an exploration of how regulations and health effects due to aviation are related.

2. Current regulations affecting emissions from aviation

Various regulations govern the environmental effects of aircraft. These standards have been set at the national and international levels and are continually updated by regulatory bodies. This chapter will give an overview of emissions regulations set by the International Civil Aviation Organization (ICAO) and the U.S. EPA. It will then discuss aviation fuels, which also influence the emissions from aircraft engines.

2.1. Overview of international emissions regulations

Aviation is regulated at international and at more local levels. Regulations for noise and emissions from aviation are analyzed, established, and updated at the international level by the International Civil Aviation Organization's Committee on Aviation Environmental Protection (CAEP) (International Civil Aviation Organization 2007b). ICAO is a body of the United Nations and consists of 190 member states from around the world. The organization was created in the United States in Chicago in 1944 as an agreement between states (known as the Chicago Convention) "in order that international civil aviation may be developed in a safe and orderly manner and that international air transport services may be established on the basis of equality of opportunity and operated soundly and economically" (International Civil Aviation Organization 1944).

ICAO regulations are agreements that member states are supposed to follow. After a standard is established or updated, ICAO's member states must submit written notifications of any differences between their standards and ICAO standards. The Chicago Convention allows disputes to be raised against member states that do not follow ICAO regulations, and states found non-compliant through the dispute process are at risk of losing their voting power in the ICAO Council and Assembly. In addition, Article 33 of the Convention states that members can derecognize aircraft airworthiness certificates from other member states that do not establish standards "equal to or above the minimum standards which may be established from time to time pursuant to" the Chicago Convention (International Civil Aviation Organization 1944).

The environmental effects of aviation gained greater attention several decades after the establishment of ICAO. At the United Nations Conference on the Human Environment in 1972, ICAO recognized that it and its member states had a responsibility to "achieve maximum compatibility between the safe and orderly development of civil aviation and the quality of the human environment" (International Civil Aviation Organization 2005). The first guidance document for the control of aviation vented fuel as well as aviation emissions (including smoke and gaseous emissions) was released by ICAO in 1977. The body created the Committee on Aircraft Engine Emissions (CAEE) in that same year, and standards for noise and emissions were proposed for an annex to the Chicago Convention in 1980 (International Civil Aviation Organization 2005). Finally, in 1983, the CAEE and ICAO's Committee on Aircraft Noise (CAN) merged to form the Committee on Aviation Environmental Protection (CAEP) (International Civil Aviation Organization 2007a).

CAEP proposes, analyzes, and updates Standards and Recommended Practices (SARPs) for the mitigation of aviation's impacts upon the environment. The SARPs for emissions are placed in Annex 16 of the Convention on International Civil Aviation, Volume II (International Civil Aviation Organization 2005, International Civil Aviation Organization 2007b). This Annex governs fuel venting as well as emissions from subsonic and supersonic turbojet and turbofan engines; the document also gives appropriate procedures for the measurement of engine emissions. The first SARPs for aircraft engine emissions were issued by ICAO in 1981 (Federal Register 1997), and the latest version of Annex 16 Volume II (Amendment 5) was released on November 24th, 2005.

2.2. ICAO emissions regulations for subsonic turbojet and turbofan engines

Annex 16 Volume II currently regulates emissions of smoke, hydrocarbons, nitrogen oxides, and carbon monoxide. Engines are regulated based on their individual manufacture date or on the manufacture date of the first production model. The regulations are written in terms of rated thrust in kiloNewtons (kN) and sometimes in

terms of rated pressure ratio. An overview of ICAO emissions regulations is shown in Figure 22, with arrows indicating regulations that affect engines or models with production dates beyond the timeline shown. The vertical lines in bars indicate where different subsets of a regulation apply to different engine years.

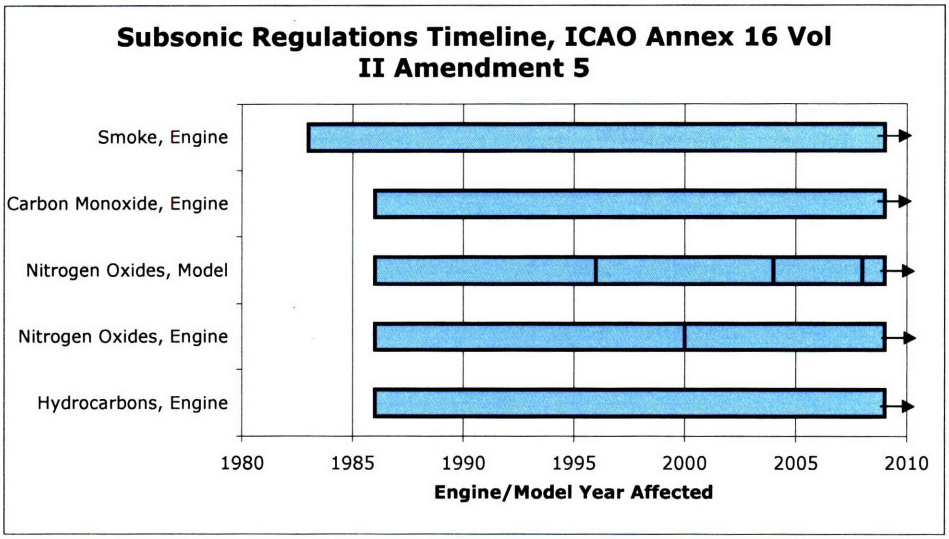

Figure 22: Overview timeline of ICAO emission regulations based on production year of individual engine or of first engine production model

2.2.1. Testing conditions

The reference atmospheric conditions that manufacturers must test their engines at are the International Standard Atmosphere conditions except that the reference absolute humidity must be 0.00634 kg of water per kg of dry air. Each engine must be tested at the four operating modes, thrust settings, and operating times shown in Table 10. The engine must be "representative of the certificated configuration," and engine loads and bleeds not necessary for engine operation do not need to be simulated (International Civil Aviation Organization 2005). The fuel used for the engine tests must meet the specifications of Appendix 4 of Annex 16 Volume II and must not contain smoke-suppressing additives (International Civil Aviation Organization 2005). Test results of engines not tested at the specified reference conditions must be corrected based on

methods described in Appendix 3 of Annex 16 Volume II (International Civil Aviation Organization 2005).

Table 10: Representative operating modes and associated thrust settings and times in mode used for subsonic engine certification testing

Operating Mode	Thrust (% Rated Thrust)	Time in Mode (Minutes)
Takeoff	100%	0.7
Climb	85%	2.2
Approach	30%	4.0
Taxi/ground idle	7%	26.0

2.2.2. Smoke Number: ICAO Annex 16 Volume II Section 2.2.2

The ICAO regulation for smoke number applies to engines manufactured on or after January 1st of 1983. The smoke number ranges from 0 to 100 and is meant to address visibility concerns; an engine is tested by passing a specific volume and mass of engine exhaust gas through a piece of filter paper and measuring the change in reflectance due to the exhaust. The allowable maximum emission of smoke per operating mode is based on the engine's rated thrust and is shown in Equation (7) as well as Figure 23 (International Civil Aviation Organization 2005):

$$SN = \min\left(83.6F_{oo}^{-0.274}, 50\right) \tag{7}$$

F_{00} is the engine's rated thrust. Note that smoke number must decrease with increasing thrust.

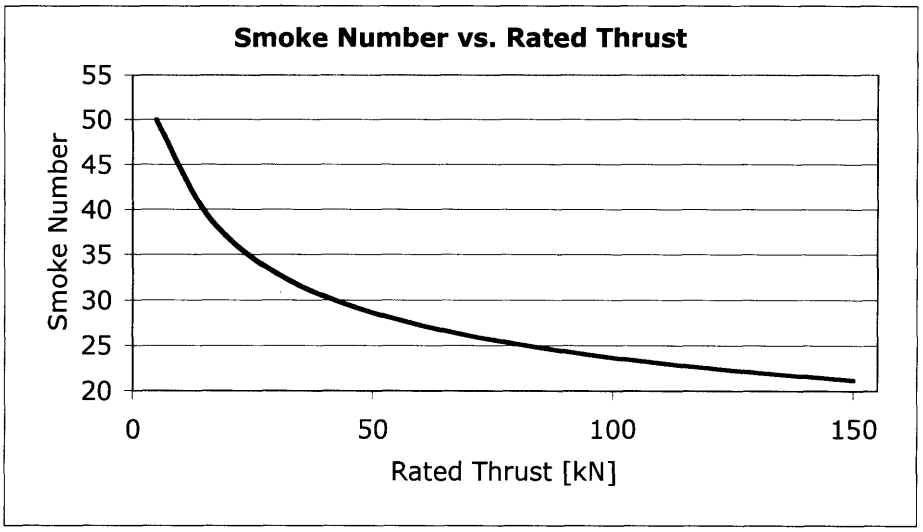

Figure 23: Maximum smoke number vs. rated thrust, ICAO Annex 16 Vol. II

2.2.3. Unburned hydrocarbons: ICAO Annex 16 Volume II Section 2.3.2

The unburned hydrocarbons regulation applies to engines of greater than 26.7 kN rated thrust manufactured on or after January 1st of 1986. An engine is tested by passing the engine exhaust through a heated flame ionization detector (FID). The allowable maximum emission of hydrocarbons is shown in Equation (8) as well as Figure 24:

$$D_p / F_{00} = 19.6 \tag{8}$$

D_p is the pollutant mass in grams. Emitted mass is allowed to increase linearly with rated thrust.

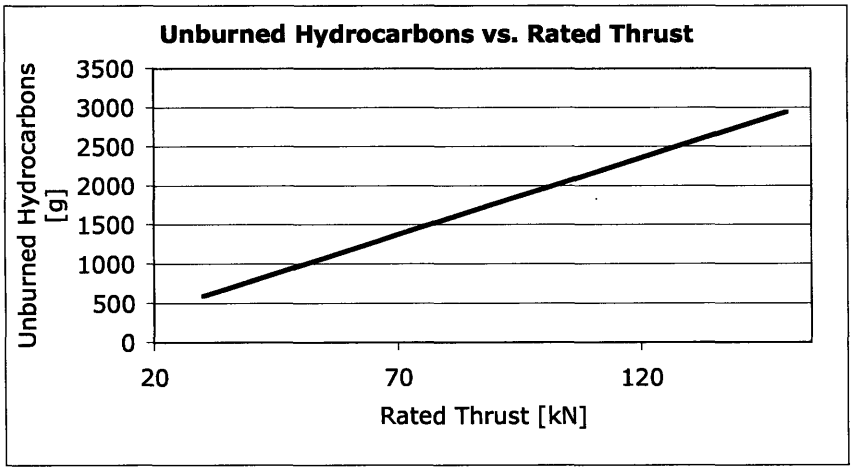

Figure 24: Maximum hydrocarbon emissions vs. rated thrust, ICAO Annex 16 Vol. II

2.2.4. Carbon monoxide: ICAO Annex 16 Volume II Section 2.3.2

The CO standard applies to engines of greater than 26.7 kN rated thrust that have been manufactured on or after January 1st of 1986. Engines are tested using a non-dispersive infrared analyzer. Allowable maximum emissions of CO are shown in Equation (9) and also in Figure 25. Like HC emissions, CO emissions are allowed to increase linearly with increasing thrust.

$$D_p/F_{00} = 118 \tag{9}$$

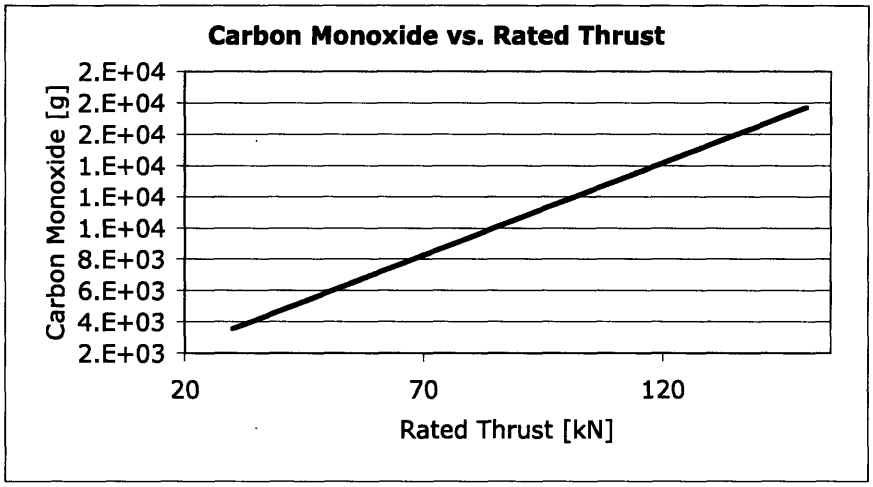

Figure 25: Maximum carbon monoxide emissions vs. rated thrust, ICAO Annex 16 Vol. II

2.2.5. Nitrogen oxides: ICAO Annex 16 Volume II Section 2.3.2.a – 2.3.2.d

ICAO's regulations for nitrogen oxides are complex and have been updated multiple times since their inception. All ICAO NO_x regulations apply to engines of greater than 26.7 kN rated thrust. The engines are tested by converting NO_2 in the exhaust to NO and then using a chemi-luminescent method where NO is reacted with ozone and the resulting radiation is measured.

The most stringent NO_x standard was set at the Sixth Meeting of CAEP (CAEP/6) and applies to engines where the first individual production model was manufactured on or after December 31st of 2007. See Equations (10) – (14) and Figure 26 for the standards.

Section 2.3.2.d.1.i: For $\pi_{00} \leq 30$ and $F_{00} > 89.0$ kN:

$$Dp/F_{00} = 16.72 + 1.4080\pi_{00} \tag{10}$$

Section 2.3.2.d.1.ii: For $\pi_{00} \leq 30$ and $26.7 < F_{00} \leq 89.0$ kN:

$$Dp/F_{00} = 38.5486 + 1.6823\pi_{00} - 0.2453F_{00} - 0.00308\pi_{00}F_{00} \tag{11}$$

Section 2.3.2.d.2.i: For $30 < \pi_{00} < 82.6$ and $F_{00} > 89.0$ kN:

$$Dp/F_{00} = -1.04 + 2.0\pi_{00} \tag{12}$$

Section 2.3.2.d.2.ii: For $30 < \pi_{00} < 82.6$ and $26.7 < F_{00} \leq 89.0$ kN:

$$Dp/F_{00} = 46.1600 + 1.4286\pi_{00} - 0.5303F_{00} + 0.00642\pi_{00}F_{00} \tag{13}$$

Section 2.3.2.d.3: For $\pi_{00} \geq 82.6$:

$$Dp/F_{00} = 32 + 1.6\pi_{00} \tag{14}$$

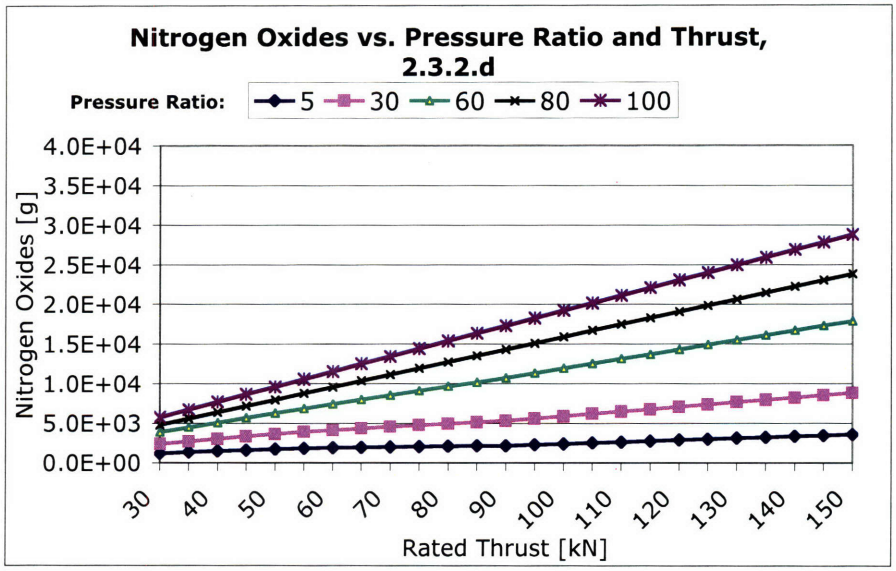

Figure 26: Maximum nitrogen oxide emissions vs. rated pressure ratio and thrust, ICAO Annex 16 Vol. II Section 2.3.2.d

A visualization of how the NO_x stringency has increased from the CAEP/4 regulations (in Section 2.3.2.c) to the CAEP/6 regulations is shown in Figure 27. It is clear from the two figures that engines with higher pressure ratios are permitted to produce more NO_x per operating mode. In particular, engines with low pressure ratios faced the biggest increase in stringency in the transition from CAEP/4 to CAEP/6 regulations.

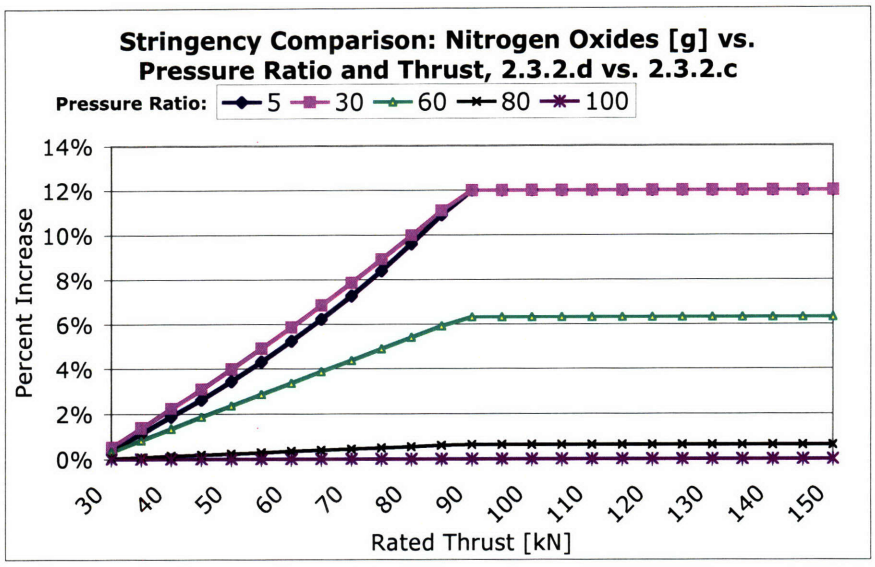

Figure 27: Visualization of percent increase in stringency from CAEP/4 (Section 2.3.2.c) to CAEP/6 (Section 2.3.2.d) NO_x regulations

2.3. Overview of emissions regulations in the United States

The United States Clean Air Act tasks the administrator of the U.S. Environmental Protection Agency with investigating aircraft emissions "in order to determine—(A) the extent to which such emissions affect air quality in air quality control regions throughout the United States, and (B) the technological feasibility of controlling such emissions" (United States Code 2005b). The Clean Air Act also states that the EPA administrator "shall, from time to time, issue proposed emission standards applicable to the emission of any air pollutant from any class or classes of aircraft engines which in his judgment causes, or contributes to, air pollution which may reasonably be anticipated to endanger public health or welfare." The Act specifies that the EPA administrator should work with the Federal Aviation Administration's administrator in the promulgation of regulations and "shall not change the aircraft engine emission standards if such change would significantly increase noise and adversely affect safety." The U.S. Secretary of Transportation must enforce compliance with the regulations (United States Code 2005c). The Clean Air Act also restricts U.S. states from taking independent action on aircraft emissions, stating that "[n]o State or political subdivision thereof may adopt or attempt to enforce any standard respecting emissions of any air pollutant from any aircraft or engine

thereof unless such standard is identical to a standard applicable to such aircraft under this part" (United States Code 2005d).

The U.S. EPA first promulgated aviation standards for fuel venting, smoke, HCs, NO_x, and CO for aircraft engines in 1973 (Federal Register 1997). In 1982, the EPA withdrew HC, CO, and NO_x standards for all newly certified aircraft gas turbine engines and also withdrew the CO and NO_x standards for new engines in model lines that were already being produced at the time of the rulemaking, leaving only smoke and fuel venting regulations. This was done because EPA analyses showed that aircraft contributed 5 ppm of CO or less to busy airports, much less than the 35 ppm 1-hour CO NAAQS of the time; EPA also deemed NO_x control technologies to be infeasible to implement in a cost-effective way. The smoke and fuel venting regulations were consistent with ICAO's 1981 SARPs, as was the HC standard that remained (Federal Register 1982).

Because the United States is a member of the International Civil Aviation Organization, changes in ICAO standards have often led to changes in U.S. regulations related to aviation emissions in order to promote the harmonization of standards (Federal Register 2005). In 1993, ICAO amended its NO_x and engine testing standards; EPA responded by adopting ICAO's 1981 NO_x and CO standards as well as ICAO's 1993 revisions to its SARPs. In 2005, EPA adopted emissions standards equivalent to ICAO's 1999 NO_x amendments as well as ICAO's 1997 test procedure amendments.

2.4. Differences between ICAO Appendix 16 Volume II Amendment 5 standards and U.S. standards

Generally, very similar standards exist between ICAO and the U.S. These standards, however, may apply to different engine or model years (Code of Federal Regulations 2005, International Civil Aviation Organization 2005). A comparative timeline is shown in Figure 28 for subsonic engines; the arrows indicate regulations that affect engines or models with production dates beyond the timeline shown. Note that EPA's regulations refer to subsonic engines using several different classes:

- Class T8: JT8D engines
- Class T3: JT3D engines
- Class TF: General turbofan engines

EPA regulations listed without classes apply to engine classes T8, T3, and TF. Most notably, EPA regulations do not include the CAEP/6 NO_x stringency standard; the regulations currently implement the regulation created at CAEP/4 and documented in Section 2.3.2.c of ICAO Annex 16 Volume II Amendment 5.

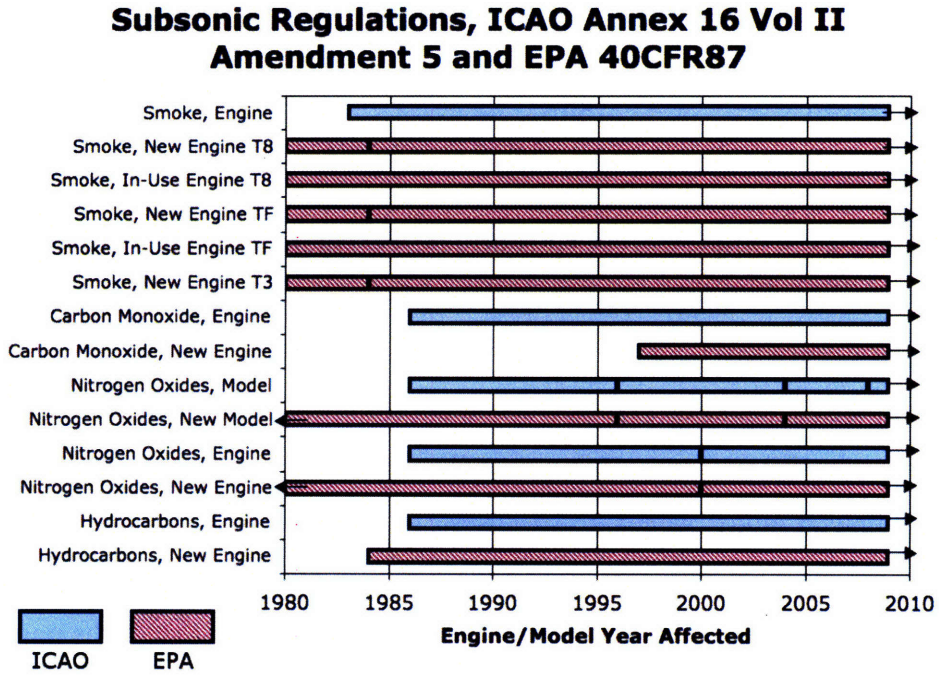

Figure 28: Comparison of ICAO and U.S. regulations

The United States regulations have a variety of smoke standards for both new and in-use engines in the three classes. A comparison of EPA and ICAO smoke number regulations is shown in Figure 29. A new engine is defined as an engine that has never been in service, while an in-use engine is defined as an engine that has been in service when a regulation goes into effect (Code of Federal Regulations 2005). EPA's regulation at 40 CFR 87.21(e)(2) applies to gas turbine engines manufactured on or after January 1st, 1984 and most closely matches the ICAO smoke number standard.

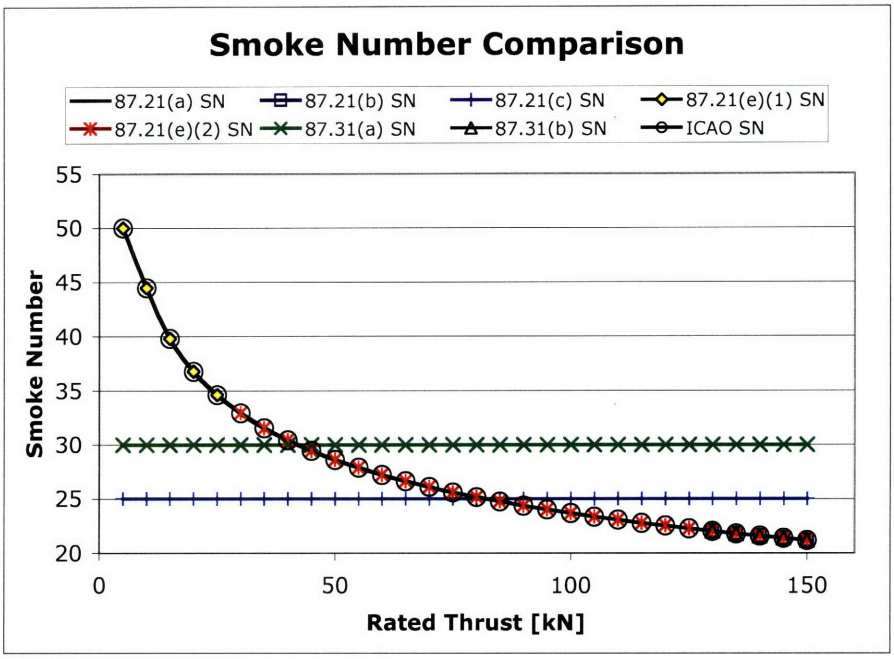

Figure 29: Comparison of EPA and ICAO smoke number regulations

Several other differences between ICAO and EPA regulations exist. For engine testing, the fuel hydrogen content by mass percentage is 13.4 to 14.1% in the United States and 13.4 to 14.3% in ICAO Annex 16 Volume II. Concerning the power setting for testing during the idle/taxi mode: Annex 16 specifies 7% thrust, but U.S. regulations specify 7% unless "the unique characteristics of an engine model undergoing certification testing at 7% would result in substantially different HC and CO emissions than if the engine model were tested at the manufacturers [sic] recommended idle power setting" (Code of Federal Regulations 2005). United States regulations also contain several exemptions from 40 CFR 87, and states cannot enforce emissions standards on exempted engines:

- Flights for short durations at infrequent intervals
- Very low production models (no more than 200 units under the same type certificate)
- Engines where the Secretary of Transportation and the EPA Administrator agree that emissions regulations are not justified based upon:

- Adverse economic impact on the engine manufacturer or aircraft and airline industries at large
- Equity issues concerning the administration of standards across all "economically competing parties"
- Public health and welfare reasons
- The infeasibility of the operator to meet certain fuel venting or smoke standards (listed at §87.11(a), §87.31(a), and §87.31(c))

In general, there are close matches between ICAO and EPA regulations for aviation emissions. Some of EPA's regulations apply to different models or years, however, and EPA has not yet promulgated an equivalent to the CAEP/6 NO_x emissions standard. The EPA regulations also include a variety of exemptions.

2.5. Aviation turbine fuels

A consideration of aviation turbine fuels is important when assessing aviation emissions because aviation fuel provides many of the compounds that undergo chemical reactions in the engine combustor and afterward to become pollutants that can be a danger to human health. There are certain ways in which jet fuels are commonly produced, and these fuels have some common properties. Petroleum aviation fuels are the most prevalent, but it is possible to make jet fuel from other compounds.

2.5.1. Petroleum fuels

Currently, almost all commercial aviation jet fuel is kerosene, which is made from petroleum (Chevron Corporation 2006). The fuels most used in commercial jet aviation are known as Jet A and Jet A-1, which are both kerosene fuels. Jet A is primarily used in the United States; it has a maximum freezing point of -40° Celsius. Jet A-1 is primarily used elsewhere, and it has a lower maximum freezing point of –47° Celsius. In the future, it may also become feasible to create and use aviation fuels derived from biomass (Massachusetts Institute of Technology and RAND Corporation 2007).

Petroleum-based aviation turbine fuel is a mixture of many types of hydrocarbon compounds and small amounts of heteroatoms (meaning "other atoms") such as sulfur and nitrogen. Petroleum itself primarily contains hydrocarbons of the paraffin, napthene, and aromatic classes. Paraffins are of the general formula C_nH_{2n+2}, where n is the number of carbon atoms and the carbon atoms are joined by single bonds; napthenes are paraffins where some of the carbon atoms are arranged in ring structures, and aromatics are hydrocarbon molecules where carbon atoms are joined by double (aromatic) bonds. Refineries convert crude petroleum into useable products using a number of processes, which can mostly be categorized into separation, upgrading, and conversion. Kerosene is obtained in a separation process when petroleum is split into various kinds of products. In the upgrading process, kerosene is treated in various ways to increase its quality. Fuel that is not treated via an upgrading process is known as "straight-run" fuel (Chevron Corporation 2006). Hydroprocessing (also called hydrotreating) is often used as an upgrading process for kerosene and involves reacting fuel with hydrogen and particular catalysts; the process of hydrotreating can be used to reduce or remove sulfur and nitrogen compounds in the fuel. Conversion is a process where large hydrocarbons are broken ("cracked") into smaller hydrocarbons; the process of hydrocracking (which is cracking using hydrogen and a catalyst) can produce diesel and kerosene products. Fuel is then distributed, usually to intermediate terminals that eventually supply the fuel to airports.

General standards for aviation turbine fuels are developed by the organization ASTM International and documented in ASTM D 1655 (ASTM International 2007). ASTM International, formerly known as the American Society for Testing and Materials, sets voluntary consensus standards for a variety of products. ASTM D 1655 standardizes various aviation fuel properties of importance, such as "energy content and combustion quality . . . stability, lubricity, fluidity, volatility, non-corrosivity, and cleanliness" (Chevron Corporation 2006).

ASTM D 1655 specifies that aviation turbine fuels must have a maximum total sulfur content of 0.30% (3000 ppm). The fuel sulfur content is important because SO_x is

created from burning sulfur-containing fuels (Flagan and Seinfeld 1988), and a substantial portion of the health impacts described in (Ratliff, Sequeira, Waitz, et al. 2008) came from SO_x-related particulate matter. The average sulfur content of aviation fuel worldwide "appears to be between 500 and 1000 ppm" (Chevron Corporation 2006). Sulfur can be removed from petroleum-based jet fuel by hydrotreating, where hydrogen reacts with fuel sulfur to form hydrogen sulfide. This hydrogen sulfide is then removed from the fuel. Hydrotreating to remove fuel sulfur, known as hydrodesulfurization (HDS), also changes other fuel properties such as energy content, aromatics content, and lubricity. For example, the U.S. EPA determined that creating diesel fuel with a sulfur content of 15 ppm would reduce volumetric energy content by "roughly 1.5 percent" (U.S. Environmental Protection Agency 2000b); creating ultra-low sulfur (ULS; 15 ppm) jet fuel would likely lead to "a similar loss" (Massachusetts Institute of Technology and RAND Corporation 2007). Energy content of fuel is described per unit mass (gravimetric) as well as per unit volume (volumetric). ULS fuel would also have a 1% lower volumetric heat of combustion, causing aircraft to need "1 percent more fuel to fly a given range" in comparison with Jet A. Furthermore, the hydrogen content in such a fuel would increase slightly in comparison with Jet A, leading to "less than a 1.5 percent increase in water emissions and a negligible decrease in CO_2 emissions."

2.5.2. Fischer-Tropsch (F-T) fuels

It is also possible to create aviation jet fuel using the Fischer-Tropsch process, which converts natural gas, coal, or other substances into liquid fuels. F-T fuels are created by converting raw ingredients (such as coal or natural gas) to "syngas" using steam and oxygen and then passing the syngas over catalysts to produce hydrocarbon products; the F-T process can theoretically create fuel from "any carbon-containing raw material" (Massachusetts Institute of Technology and RAND Corporation 2007). F-T fuels are low in sulfur and contaminants. The fuels' mass-based energy content is "2 percent higher than that of Jet A," while the volumetric energy content is "3 percent lower." It is already feasible to produce F-T fuel from natural gas, and F-T fuels from this source have "essentially no sulfur or aromatics" (Chevron Corporation 2006).

2.6. Review

This chapter assessed the current regulations that govern emissions from aircraft at the national and international levels. International regulations are set by the International Civil Aviation Organization, which is a body of the United Nations. ICAO's Committee on Aviation Environmental Protection investigates the effects of regulations and develops or changes its Standards and Recommended Practices accordingly. Currently, ICAO regulations address emissions of smoke, NO_x, CO, and HCs from engines designed for subsonic operation. There are also SARPs that dictate testing procedures for engine certification. Emissions regulations in the United States closely align with regulations set by ICAO. There are a few notable differences; in particular, the EPA has not promulgated a version of the CAEP/6 NO_x standard (EPA regulations currently implement the CAEP/4 NO_x standard). EPA also has several smoke number standards for specific jet engine classes as well as a list of situations where manufacturers are exempt from emissions requirements (such as for small production runs).

Finally, this chapter discussed aviation fuels. An understanding of fuels is important because many of the emissions from aviation are related to the content of various compounds in aviation fuel. Fuel sulfur content, in particular, is important. Almost all aviation fuel is kerosene, which is created from petroleum. Products of petroleum can be processed in various ways to produce aviation fuel with certain properties, such as higher hydrogen content or lower sulfur content. EPA has already mandated the production of ultra low-sulfur diesel fuel, which may make production of low-sulfur jet fuel more feasible. It is also possible to produce fuel from non-petroleum sources using the Fischer-Tropsch process. The next chapter will connect aviation emission regulations with the human health effects of aviation emissions.

3. An assessment of the health implications of aviation emissions regulations

Chapters 1 and 2 of this thesis have discussed how aviation affects human health through perturbations in air quality and what standards govern aviation emissions. The link from emissions through dispersion and transformation and ultimately to human exposure has been explored. Chapter 3 attempts to assess this link in the context of current aviation emissions regulations to illuminate an impact pathway from regulations to human health effects. It will then become possible to explore what kinds of regulatory strategies could effectively mitigate human health impacts from aviation. A linear model is developed as an assessment tool. Then, three strategies are briefly analyzed and presented: two strategies addressing the deployment of ultra low-sulfur fuel, and one strategy analysis performed by ICAO CAEP's Forecasting and Economic Analysis Support Group to address an increase in the stringency of ICAO's NO_x regulations.

3.1. The connection between regulations and health impacts

The regulations that govern aviation emissions ultimately affect human health incidences due to pollutant exposure. A visualization of this impact pathway is shown in Figure 30 for EPA and ICAO regulations as well as the fuel sulfur standard. The impact pathway shows how regulations ultimately connect with human health incidences via aircraft operations and the dispersion and transformation of aircraft emissions as well as human exposure to them. For example, changes in smoke number standards cause changes in the mass of primary PM from aircraft operations, which ultimately causes changes in health incidences due to primary PM exposure. Each of the standards mentioned in the impact pathway will be discussed in turn.

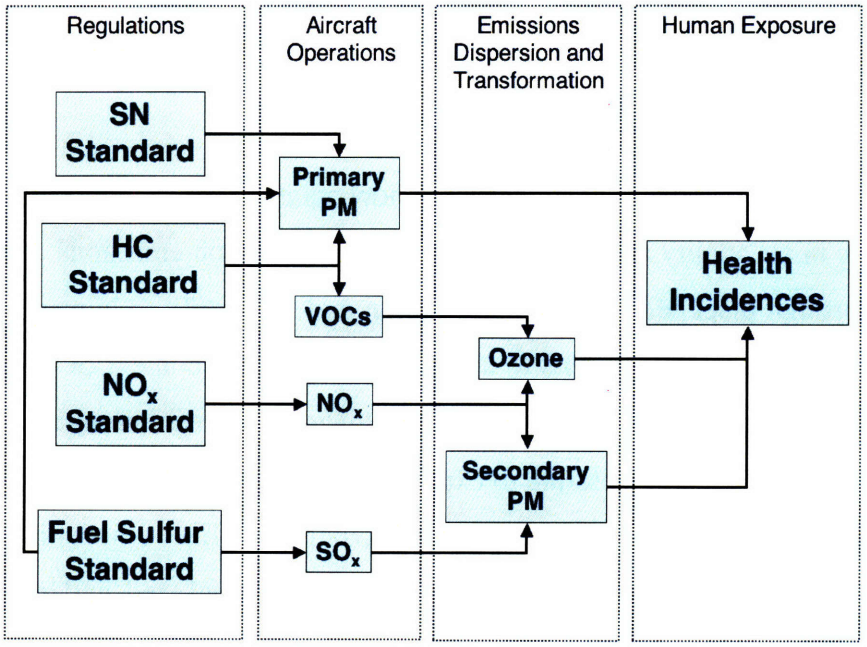

Figure 30: Pathway from emissions standards to health impacts

3.1.1. Smoke number standard

The smoke number standard was originally designed to address visibility concerns, and smoke number is determined using a measurement system that detects changes in the reflectance of a filter exposed to engine exhaust (International Civil Aviation Organization 2005). Smoke number, however, can be correlated with the formation of nonvolatile particulate matter, as has been done in the FOA3 and FOA3a particulate matter models (Ratliff, Sequeira, Waitz, et al. 2008). ICAO and EPA smoke number standards have not changed since the early 1980's. They generally allow smoke to increase with decreases in rated thrust, except for the JT3D and JT8D engine classes in EPA regulations (Code of Federal Regulations 2005).

Nonvolatile PM accounted for approximately 1% or less of the total emitted mass from aircraft in the Energy Policy Act and RSM inventories, as shown in Figure 31.

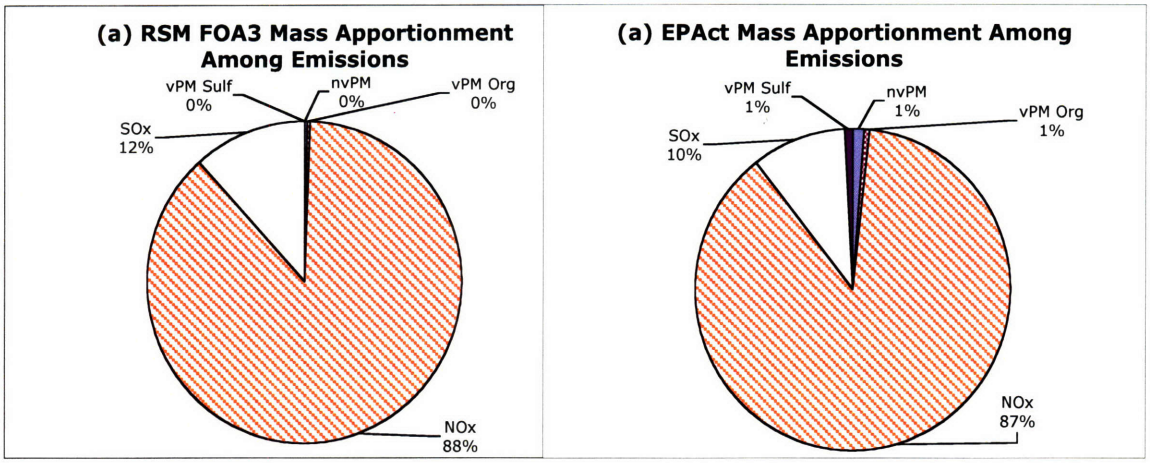

Figure 31: Apportionment of total emitted mass in the Energy Policy Act and RSM inventories

Because nonvolatile PM is non-reactive, it reaches ground-level particulate matter monitors in the same chemical form in which it was emitted. As shown in Figure 14, nonvolatile PM emissions accounted for approximately $140 in health costs per kilogram (or $140,000 per tonne) due to PM-related premature mortality of adults age 30 and over. Nonvolatile PM was the second most damaging emission per unit mass. Because the emitted mass of volatile PM was small in the relative sense, nonvolatile PM was responsible for the least total health costs. A similar message can be drawn for the RSM inventory.

3.1.2. Unburned hydrocarbons standard

The unburned hydrocarbons standard connects with PM-related human health impacts due to particulate matter because unburned hydrocarbons form a portion of the volatile primary PM mass. Unburned hydrocarbons also affect health impacts due to ozone exposure because of their relationship to ambient VOCs, which affects ozone production (U.S. Environmental Protection Agency 2006b, Wey, Anderson, Wey, Miake-Lye, Whitefield and Howard 2007). The correlation between hydrocarbon emissions and volatile particulate matter is important, as some hydrocarbon species begin to condense to particles by 30 meters downstream of the engine exit. This correlation is approximated in FOA3 and FOA3a. HC-related volatile particles disperse in the atmosphere and are measured in PM monitors as organic carbon. The hydrocarbon standard has not been

changed in the United States or at the international level since its creation (Code of Federal Regulations 2005, International Civil Aviation Organization 2005).

The knowledge of how primary PM emissions correlate with fuel composition and engine combustor design is still at an early stage. Researchers are not entirely certain how hydrocarbon emissions at the engine nozzle influence the creation of volatile primary particulate matter farther downstream in the exhaust plume. FOA3a assumes that the mass of volatile primary PM from hydrocarbons scales directly with the hydrocarbon emissions of the engine. Based on the assumptions of FOA3a, emissions of volatile primary PM from hydrocarbons was 40% of the total volatile primary PM in the Energy Policy Act inventory and was responsible for 19% of the health incidences; see Figure 32. This gave volatile primary PM from hydrocarbons a marginal damage of approximately $340,000 per tonne, the highest of all PM species. In comparison, organics-related volatile primary PM was 66% of the total volatile primary PM in the RSM FOA3 inventory and had approximately the same marginal damages.

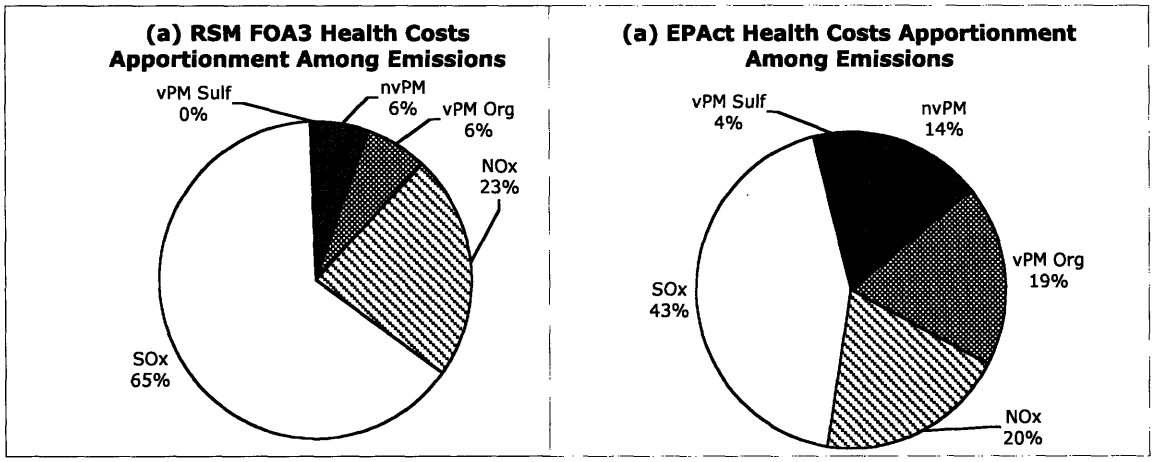

Figure 32: Health cost apportionment among emissions by percentage for the Energy Policy Act and RSM inventories

3.1.3. NO_x standard

The NO_x standard connects with human health impacts through the influence of NO_x on concentrations of secondary ammonium nitrate and ozone in the atmosphere. NO_x is

created when the nitrogen and oxygen in air experience high temperatures during the combustion of fuel in the engine's combustor. More NO_x is created at higher combustion temperatures; thus, the amount of NO_x created is closely related to engine properties such as pressure ratio and combustor design (International Civil Aviation Organization 2004). Over timescales of minutes to days, NO_x reacts with ammonia in the atmosphere and becomes ammonium nitrate, which reaches ground level as fine particulate matter and can be inhaled by humans. Based on the Energy Policy Act and RSM inventories, NO_x emissions had the lowest marginal damage per unit mass: approximately $2 per kilogram or $2,000 per tonne, as shown in Table 7, Figure 14, and Figure 15. Because NO_x emissions made up 87% of the total aircraft emissions, as shown in Figure 31, NO_x emissions were ultimately responsible for 20% of the total health costs due to aviation emissions in the Energy Policy Act inventory. Based on the RSM inventory, NO_x emissions were 88% of total emitted mass and caused 23% of the health costs. Ambient concentrations of ammonium nitrate from aviation emissions of NO_x were responsible for 18% of the health costs due to aviation-related ambient PM concentrations based on the Energy Policy Act inventory and 20% based on the RSM inventory; see Figure 13.

The NO_x standard has changed several times at the international and United States levels since its creation, and the current ICAO and EPA standards allow engines with larger pressure ratios to produce more NO_x. See Section 2.2.5 and Section 2.3 of this thesis for further details. The most stringent increase at the international level targeted engines produced after December 31st of 1995 vs. engines produced before that date (International Civil Aviation Organization 2005). In comparison, further stringency increases have been smaller.

3.1.4. Fuel sulfur standard

Fuel sulfur is important in the impact pathway because it causes health effects by influencing volatile primary particulate matter formation and the formation of secondary ammonium sulfate particles. Volatile particles cannot be found immediately at the exit plane of a jet engine, but they compose a large fraction of the particles in an exhaust plume measured 30m from the engine exit nozzle due to condensation of volatile gases.

The current aviation gas turbine fuel sulfur standard is 3000 ppm (ASTM International 2007). However, measured fuel sulfur levels vary by location, and the weighted mean sulfur level of aviation fuels acquired by the U.S. military was 776 ppm in 2006. This mean is potentially indicative of commercial aviation fuel sulfur levels (U.S. Environmental Protection Agency 2004c).

Approximately 60% of the total volatile primary PM mass in the Energy Policy Act inventory was due to the sulfur in fuel. Emissions of volatile primary PM from sulfur caused 4% of the total health costs (with total volatile primary PM emissions being responsible for 23%). For the RSM inventory, sulfate-related volatile primary PM composed 33% of total volatile primary PM emitted mass and was responsible for 0% of the health costs. The relative apportionment of health costs to aviation emissions is shown in Figure 32 for both inventories. Recall that the mass of sulfur-related primary PM in the Energy Policy Act inventory was predicted using a fuel sulfur assumption of 680 ppm, but 78 of the 325 airports in the Energy Policy Act study were incorrectly assigned a fuel sulfur assumption of 400 ppm. Volatile primary PM from fuel sulfur content had a marginal damage of approximately $47,000 per tonne across both inventories.

Fuel sulfur also has a very important role in the creation of SO_x emissions, which were 10% of the Energy Policy Act inventory by mass (Figure 31). SO_x is emitted by aircraft and becomes ammonium sulfate particles over timescales of minutes to days. SO_x will also preferentially bond with ammonia in the atmosphere, reducing the amount of ammonia available for the formation of ammonium nitrate.

SO_x emissions caused damages of approximately $43,000 to $46,000 per tonne across the Energy Policy Act and RSM inventories. The amount of SO_x mass produced, however, caused SO_x emissions to dominate the total health costs. For the Energy Policy Act inventory, SO_x emissions were responsible for 43% of the incidences of PM-related premature mortality, while ammonium sulfate concentrations from SO_x represented 46% of the total health costs related to aviation-induced PM concentrations across the

continental United States and 46% of the premature mortality incidences in Los Angeles County. Recall that Los Angeles County experienced 18% of the continental U.S.-wide health impacts from PM mortality. For the RSM inventory, SO_x emissions were responsible for 65% of the health costs, while ammonium sulfate concentrations from SO_x represented 69% of the total aviation-related health costs due to PM exposure. 18% of the total premature mortality incidences were in Los Angeles County, where 66% of the mortality incidences were caused by ammonium sulfate concentrations from SO_x. See Figure 17 for the apportionment of incidences to concentrations of PM species in Los Angeles County.

3.2. Potential areas of policy focus

Understanding the health costs of each emitted pollutant and the resulting ambient concentration illuminates opportunities for policymakers to address the health effects of aviation due to air quality changes. The Energy Policy Act study indicated that premature mortality of adults age 30 and over due to $PM_{2.5}$ exposure represented 97% of the health costs, as discussed in Section 1.4.6 of this thesis. In contrast, health costs from ozone exposure were negative due to ozone disbenefits, and the magnitude of ozone health costs was a small fraction of the magnitude of $PM_{2.5}$ health costs. For this reason, policies addressing the health effects of aviation emissions could be most effective if they focus on $PM_{2.5}$ emissions first.

3.2.1. Nonvolatile primary PM

Nonvolatile primary PM emissions could be addressed through changes in engine technology, as combustor technology strongly influences nonvolatile primary PM formation. It is recognized that increasing mixing in the engine's combustor region decreases the formation of nonvolatile particles, and increasing the air-fuel ratio in the combustor can decrease or increase the formation of such particles (Blevins 2003). Nonvolatile primary PM production is also influenced by the type of fuel injection system; for example, explorations of various fuel injection systems in the 1970's demonstrated that using airblast atomizers instead of pressure atomizers reduced smoke substantially.

Changes in technology, however, will require manufacturers to invest money to develop the new technology. Airlines must then purchase the new engines and retire their old ones, which will take time. Policymakers may be able to address nonvolatile primary particulate matter relatively quickly by reducing the aromatics content in fuel, since fuels with high aromatics content produce more nonvolatile particulate matter when combusted (Chevron Corporation 2006).

3.2.2. Hydrocarbons

Volatile particulate matter dominates primary PM mass in the engine plume at 30 meters from the engine nozzle. Hydrocarbons comprised 40% and 66% of the mass of volatile primary PM emissions in the Energy Policy Act and RSM inventories, respectively. Because of this, reducing hydrocarbon emissions would reduce primary particulate matter concentrations, leading to health benefits. Reducing hydrocarbon emissions can also reduce ambient VOC concentrations, but this may or may not reduce ozone due to potential disbenefits arising from the complex chemical processes that create ozone. A dispersion model that includes meteorology, other secondary pollutants such as NO_x, sunlight, and other factors would have to be executed to determine the nature of a reduction in VOCs (U.S. Environmental Protection Agency 2006b). However, because health costs due to ozone were of a small magnitude in the Energy Policy Act study, the effect of a hydrocarbon emissions standard on ozone concentrations may be of less importance to policymakers.

3.2.3. NO_x

The reduction of NO_x will reduce human health impacts caused by secondary ammonium nitrate formation. Reducing NO_x may or may not decrease ozone-related human health effects due to complex chemical processes. The NO_x standards have changed numerous times at the ICAO and EPA levels since they were enacted, and the finding that NO_x emissions were responsible for 20% of the health costs in the Energy Policy Act inventory and 23% of the costs in the RSM inventory suggests that a continued focus on NO_x is important.

Unfortunately, increasing NO_x stringency will require changes in engine technology, potentially costing manufacturers and airlines substantial resources, and time is required for the development of new engine technology and gradual uptake in the global commercial aircraft fleet. Because engine fuel efficiency increases with increasing pressure ratio, reductions of engine pressure ratio to control NO_x formation can cause increases in CO_2 production. This leads to a well-known tradeoff between NO_x and CO_2 emissions. For a discussion of ICAO CAEP's assessment of a proposed NO_x stringency, see Section 3.3.4 of this thesis.

3.2.4. Sulfates

Fuel sulfur is responsible for volatile primary PM from sulfate as well as SO_x emissions. Together, these emissions accounted for 47% of the emissions-related health costs in the Energy Policy Act inventory and 65% of the emissions-related costs in the RSM inventory. It is possible to remove sulfur from petroleum-based jet fuel using the process of hydrodesulfurization (HDS); this process, however, also changes other fuel properties (Chevron Corporation 2006, Massachusetts Institute of Technology and RAND Corporation 2007). A switch to a lower-sulfur fuel would likely not require substantial changes in aircraft or engine technology. Production of such a fuel would require changes in refineries, but as part of its Clean Air Nonroad Diesel Rule, EPA has already mandated a reduction of sulfur in diesel fuel to 15 ppm (U.S. Environmental Protection Agency 2004c). This has spurred refineries to begin producing 15 ppm diesel fuel.

Because the health effects of sulfates could be reduced just by switching aviation fuels, addressing aviation fuel sulfur content could be an effective way to address aviation-related health impacts due to air quality changes. It is also possible, however, for reductions in SO_x emissions to allow more ammonium nitrate to form due to the bounceback effect. The effects of fuel sulfur reductions are explored in Section 3.3.3 of this thesis.

3.3. Assessment of policy options

3.3.1. Analysis methodology

To explore the effects of a few policy options, a model was developed using the assumption that health impacts from aviation scale linearly with aviation emissions. As an input, the model takes an emissions inventory that results from a change in policy. The model utilizes data from the Energy Policy Act study (aviation emissions, aviation-induced perturbations in pollutant concentrations, and corresponding health incidences and costs) as a calibrating dataset and predicts health impacts by comparing the input emissions inventory with the Energy Policy Act inventory. This model was also used to compare the Energy Policy Act inventory with the RSM FOA3 inventory in this thesis.

Because premature mortality from $PM_{2.5}$ dominated the health costs in the Energy Policy Act study, the model focuses on $PM_{2.5}$ concentrations and health impacts. The CRF of interest is (Pope, Burnett, Thun, et al. 2002), used in the EPA's final regulatory analysis of non-road diesel emissions controls (U.S. Environmental Protection Agency 2004d), the 2006 revisions to the National Ambient Air Quality Standards for particulate matter (U.S. Environmental Protection Agency 2006a), and the 2005 final Clean Air Interstate Rule regulatory impact analysis (U.S. Environmental Protection Agency 2005b). BenMAP results based on this CRF for the Energy Policy Act study indicated that premature mortality from $PM_{2.5}$ for adults age 30 and over accounted for approximately 97% of the health costs, as shown in Section 1.4.6 of this thesis.

The model incorporates a number of assumptions to predict health effects. The primary assumption is that health impacts due to aviation scale linearly with aviation's perturbation of ambient $PM_{2.5}$ concentrations. This assumption was made because the perturbations of $PM_{2.5}$ and ozone ambient concentrations due to aviation activity are relatively small. To explore the linear assumption, PM concentrations from the Energy Policy Act study were scaled using several factors (0.5, 1, 1.5, 2, 5) and re-processed using BenMAP. Point health incidences from the (Pope, Burnett, Thun, et al. 2002) CRF

given by BenMAP are shown vs. the concentrations scaling factor in Figure 33 with a superimposed trend line, supporting an assumption of linearity.

As a further exploration, the point health incidences given by the scaled concentrations were divided by the number of health incidences resulting from the original Energy Policy Act study (in other words, the un-scaled health incidences) as shown in Equation (15) and plotted in Figure 34.

$$S_P = \frac{y_S}{y_0} \tag{15}$$

y_S is the number of point health incidences resulting from the scaled PM concentrations, and y_0 is the number of point incidences due to the original (un-scaled) PM concentrations in the Energy Policy Act study. Figure 34 also supports an assumption of linearity.

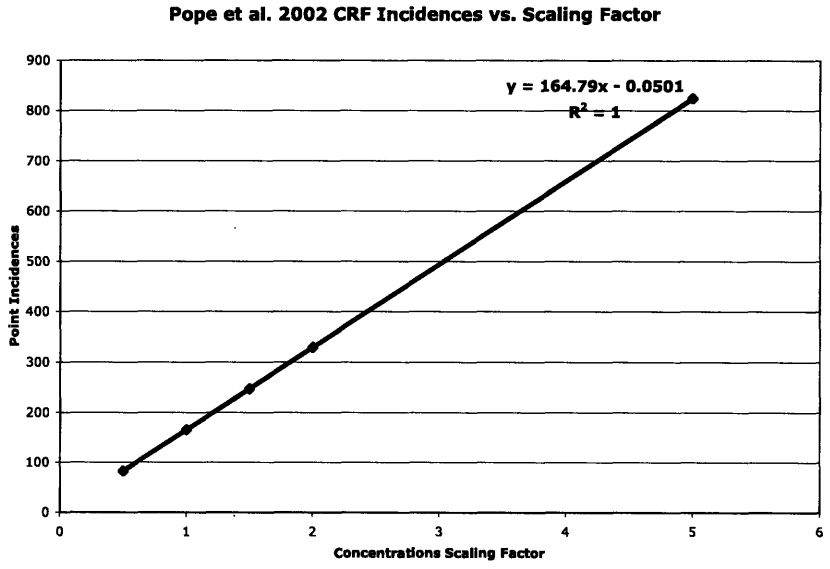

Figure 33: Scaling of Pope et al. 2002 CRF results with scaling in ambient PM concentration perturbations due to aviation

The model assumes that the linear assumption holds true at the county level as well. As shown in Section 1.4.6 of this thesis, aviation was responsible for less than 1% of the

$PM_{2.5}$ concentrations in all counties analyzed as part of the Energy Policy Act study, suggesting the acceptability of a small perturbation assumption.

In addition, the model assumes that point health costs scale with PM concentrations in the same way that the point health incidences scale. The breakdown of total health effects to the county level showed that incidence and cost percentages were essentially the same, supporting this assumption; see Section 1.4.6 of this thesis. Also, it is assumed that each species of PM has the same health impact per unit mass inhaled. The health effects of specific PM species are still being investigated by researchers, but this assumption has been used in such studies as (Levy, Wilson, Evans and Spengler 2003) and (Rojo 2007). BenMAP itself only accepts total $PM_{2.5}$ concentrations as input. Because of this, the linear scaling assumption was not tested for individual PM species concentrations.

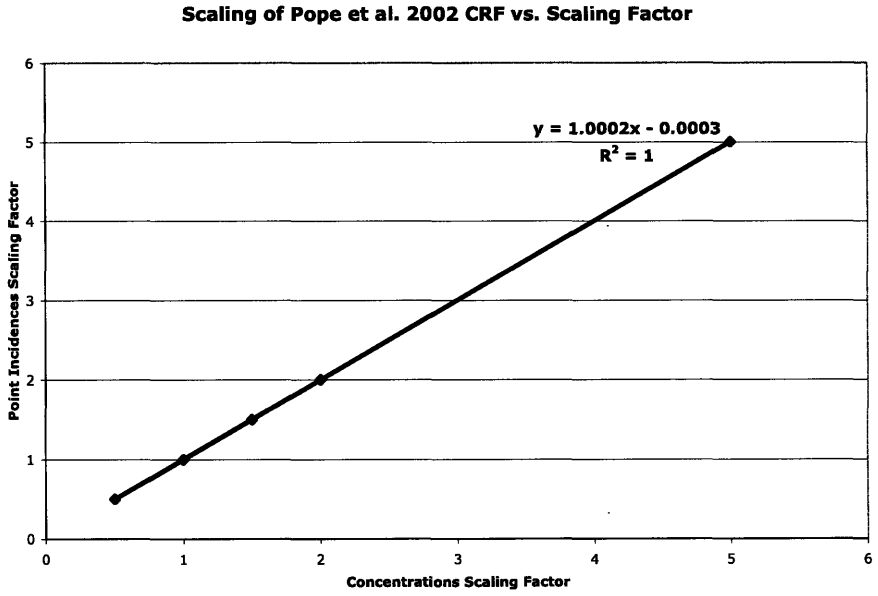

Figure 34: Scaling of Pope et al. 2002 CRF for premature mortality from $PM_{2.5}$ vs. scaling of aviation-related $PM_{2.5}$ concentrations

Based on the assumptions above, a linear scaling between health effects and PM concentrations was derived as shown in Equations (16) and (17):

$$\frac{y_S}{y_0} \approx \frac{C_{PM_{total}S}}{C_{PM_{total}0}} \Rightarrow S_P \approx S_C \tag{16}$$

$$\frac{y_{HC_S}}{y_{HC_0}} \approx \frac{C_{PM_{total}S}}{C_{PM_{total}0}} \Rightarrow S_{HC} \approx S_C \tag{17}$$

$C_{PM_{total}0}$ and $C_{PM_{total}S}$ are the original and scaled total concentrations of aviation-related particulate matter, respectively, and S_C is the scaling factor used to scale the $PM_{2.5}$ concentrations due to aviation as determined in the Energy Policy Act study. y_{HC_0} and y_{HC_S} are the original and scaled health costs, with S_{HC} being the corresponding scaling factor. In essence, the relation states that a doubling of aviation-related PM concentrations in a county leads to a doubling in aviation-related health incidences and health costs in that county. This assumes that the population and population breathing rates in a county remain constant.

To apply the scaling relations, the relationship between the input emissions inventory and $C_{PM_{total}S}$ must be determined. In the model, the total aviation-related PM concentration due to the input aviation emissions inventory is taken to be the sum of primary volatile and nonvolatile PM, ammonium nitrate, and ammonium sulfate concentrations, as shown in Equation (18).

$$C_{PM_{total}S} = C_{PM_{nvpri}S} + C_{PM_{org}S} + C_{PM_{AN}S} + C_{PM_{AS\_SOx}S} + C_{PM_{AS\_Sulf}S} \tag{18}$$

$C_{PM_{total}S}$ is the total concentration of particulate matter, $C_{PM_{nvpri}S}$ and $C_{PM_{org}S}$ are the respective nonvolatile and organics-related volatile primary PM components, $C_{PM_{AN}S}$ is the ammonium nitrate component of secondary PM, $C_{PM_{AS\_SOx}S}$ is the ammonium sulfate due to SO_x emissions, and $C_{PM_{AS\_Sulf}S}$ is the ammonium sulfate concentration due to sulfate-related volatile primary PM emissions. This relation may lead to a downward bias in concentrations, as there are other kinds of secondary particulate matter in addition to ammonium nitrate and ammonium sulfate (Rojo 2007, U.S. Environmental Protection Agency 2004a). For example, volatile primary PM from fuel organic compounds is

measurable by 30 meters behind a jet engine's exit, but not all organic molecules may have condensed by then.

A key assumption of the model is that there is a linear scaling between primary PM emissions from aviation in a county and primary PM concentrations in that same county. According to (Greco, Wilson, Spengler and Levy 2007), however, emissions in a county are likely to lead to changes in pollutant concentrations hundreds of kilometers away. To deal with this, the linear model first computes new PM concentrations in counties with aviation emissions. It then compares the new concentrations to the PM concentrations determined in the Energy Policy Act study to derive average concentration scaling factors across the counties with emissions. The model uses these factors to scale PM concentrations in counties without aviation emissions. This implements a rough assumption that PM concentrations in counties without aviation emissions scale similarly with counties that do have aviation activity when an emissions inventory changes. For example, if average concentrations of ammonium sulfate are found to increase by 4% across the counties with emissions inventories, ammonium sulfate concentrations in the counties without emissions inventories are assumed to increase by the same factor.

To obtain nonvolatile and organics-related volatile primary PM concentrations in counties with emissions, Equations (19) and (20) are used:

$$C_{PM_{nvpri}S} \approx T_{nvpri} E_{PM_{nvpri}S} \tag{19}$$

$$C_{PM_{org}S} \approx T_{org} E_{PM_{vpri\_org}S} \tag{20}$$

$C_{PM_{nvpri}S}$ is a concentration of primary nonvolatile particulate matter, and $E_{PM_{nvpri}S}$ is a mass of nonvolatile primary particulate matter emissions from the input inventory. The same relationship can be drawn for organics-related volatile primary particulate matter. T_{nvpri} and T_{org} represent linear factors describing dispersion: for every unit of emissions produced in a county, concentrations change by a factor T $\mu g/m^3$ in that same county. These factors are computed in advance using emissions and concentrations from the

Energy Policy Act study and the same Equations (19) and (20); they are then used to determine the effects of the input inventory. The implicit assumption is that these dispersion factors remain constant.

The same linear scaling assumption is drawn between sulfate-related emissions from aviation in a county and secondary ammonium sulfate concentrations in that county, as in Equation (21). Recall that both SO_x emissions and sulfate-related volatile primary PM emissions will undergo chemical transformation processes in the atmosphere and ultimately be measured in FRM monitors as sulfates, from which ammonium sulfate concentrations can be derived.

$$C_{PM_{AS}S} \approx T_{AS}\left(E_{SOxS} + E_{vPM\_SulfS}\right) \tag{21}$$

$C_{PM_{AS}S}$ is the concentration of ammonium sulfate, E_{SOxS} is the emitted mass of sulfur oxides, and E_{vPM_SulfS} is the emitted mass of sulfate-related volatile primary PM. T_{AS} is the dispersion factor relating emissions of sulfates and SO_x to concentrations of ammonium sulfate; it is computed in advance using Energy Policy Act data. To apportion ammonium sulfate concentrations individually to SO_x and sulfate-related volatile primary PM emissions, the relative proportions of each mass of emissions are used, as shown in Equations (22) and (23):

$$C_{PM_{AS\_SOx}S} = C_{PM_{AS}S}\frac{E_{SOx}}{E_{SOxS} + E_{vPM\_SulfS}} \tag{22}$$

$$C_{PM_{AS\_Sulf}S} = C_{PM_{AS}S}\frac{E_{vPM\_SulfS}}{E_{SOxS} + E_{vPM\_SulfS}} \tag{23}$$

Because ammonia preferentially bonds with sulfate in the atmosphere (reducing the amount of ammonia available to create ammonium nitrate), the computation of ammonium nitrate concentrations must include the bounceback effect, as shown in Equation (24):

$$C_{PM_{AN}S} \approx T_{AN} E_{NOxS} + T_{BB}\left(E_{SOxS} + E_{vPM\_SulfS}\right) \quad (24)$$

$C_{PM_{AN}S}$ is the concentration of ammonium nitrate. T_{AN} is the dispersion factor relating emissions of NO_x to concentrations of ammonium nitrate without the bounceback effect, while T_{BB} describes the bounceback effect itself. It is not possible to easily determine the magnitude of the bounceback effect by looking at Energy Policy Act study data because the final dataset is the result of a complex atmospheric chemistry model, so the magnitude of the bounceback effect was assumed for the model based on intake fraction coefficients. An intake fraction relates emissions of a certain compound to human exposure to a specific compound; these two compounds need not be the same (Greco, Wilson, Spengler and Levy 2007).

The model uses two intake fractions in (Rojo 2007): an intake fraction relating SO_x emissions to the bounceback effect, and an intake fraction relating SO_x emissions to ammonium sulfate concentrations. The assumption is that the ratio between T_{BB} and T_{AS} is the same as the ratio between the bounceback intake fraction and the ammonium sulfate intake fraction, as shown in Equation (25).

$$\frac{T_{BB}}{T_{AS}} \approx \frac{iF_{BB}}{iF_{AS}} \quad (25)$$

Based on (Rojo 2007), the ratio is approximately -0.13. In other words, if a specific change in SO_x emissions causes ammonium sulfate concentrations to increase by 0.1 $\mu g/m^3$, ammonium nitrate concentrations will decrease by 0.013 $\mu g/m^3$ if the NO_x emissions inventory remains the same. The implicit assumption is that population exposure and breathing rates remain constant. Recall that the bounceback effect is a nonlinear process and has been linearized for the model.

Now the new total concentration of PM resulting from the input emissions inventory can be computed using Equation (18). This is done for every county with aircraft emissions. The model then uses Energy Policy Act PM concentrations to compute average factor

changes in PM concentrations across all counties with aircraft emissions and applies these average scaling factors to counties without aircraft emissions. This serves as a rough model of physical transport in the atmosphere.

Finally, the new total health incidences and costs can be computed for every county (with and without aviation emissions) using Equations (16) and (17). Due to the assumption that all PM has the same health impact per unit mass inhaled, the apportionment of health incidences and costs use the same percentages as the apportionment of concentrations among various PM species. For example, if 20% of the total aviation-related PM concentration in a county is ammonium nitrate, 20% of health incidences in that county will be assigned to ammonium nitrate.

3.3.2. Effects of a continental U.S.-wide fuel sulfur stringency increase

It has been determined in other parts of this thesis that the reduction of sulfur in fuel could contribute significantly to the mitigation of aviation-related health effects. To explore the impacts of such a reduction, an ultra-low sulfur (ULS, or 15 ppm) fuel emissions inventory was assessed with the linear model. The inventory of ULS fuel was assembled by James Hileman at the Massachusetts Institute of Technology and was based on FAA's AEDT/SAGE inventory (Federal Aviation Administration 2006) from the year 2005. Primary PM emissions inventories were modeled with FOA3.

For a fair comparison, an AEDT/SAGE inventory resulting from a fuel sulfur content of 600 ppm was also assessed with the linear model. It was decided to use this inventory instead of the Energy Policy Act inventory because the Energy Policy Act inventory has many differences in comparison with the AEDT/SAGE 600 ppm inventory. Both AEDT inventories contain 515 airports representing 243 of the 273 counties with airports in the Energy Policy Act study. The results of the assessment are shown in Table 11 and Figure 35 and as well as Table 12 and Figure 36.

Table 11: Comparison of health incidences caused by the AEDT 600 ppm and ultra low-sulfur fuel inventories

Inventory	Nonvolatile Primary PM	vPM Organics	Ammonium Nitrate	SO_x Ammonium Sulfate	vPM Ammonium Sulfate
600 ppm	5	6	30	65	0
ULS	5	6	53	2	0
Ratio	*1.0*	*1.0*	*0.6*	*40.1*	*40.1*

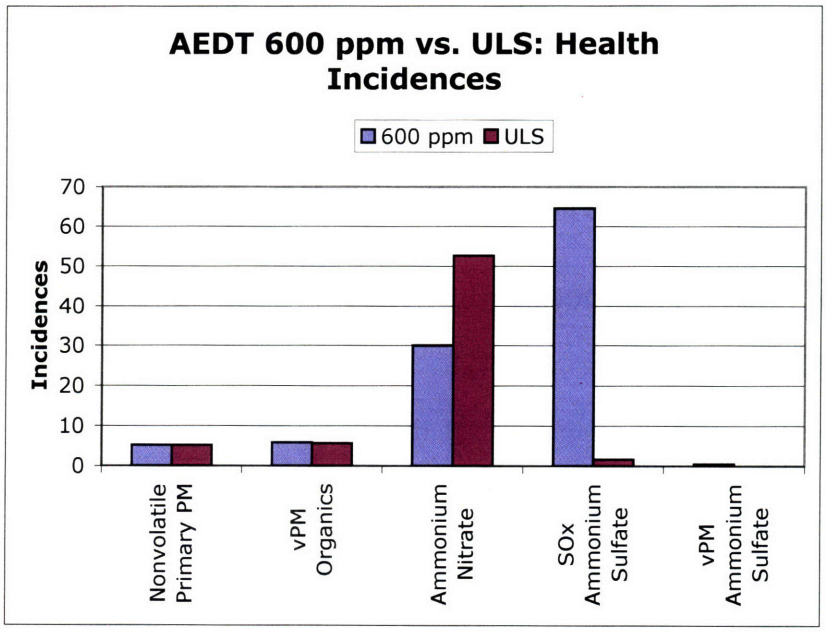

Figure 35: Comparison of the AEDT/SAGE 600 ppm fuel sulfur inventory with an inventory based on a fuel sulfur content of 15 ppm

Table 12: Comparison of health costs (in millions) caused by the AEDT 600 ppm and ultra low-sulfur fuel inventories

Inventory	Nonvolatile Primary PM	vPM Organics	Ammonium Nitrate	SOx Ammonium Sulfate	vPM Ammonium Sulfate
600 ppm	$28	$31	$161	$345	$3
ULS	$28	$31	$281	$9	$0
Ratio	*1.0*	*1.0*	*0.6*	*40.1*	*40.1*

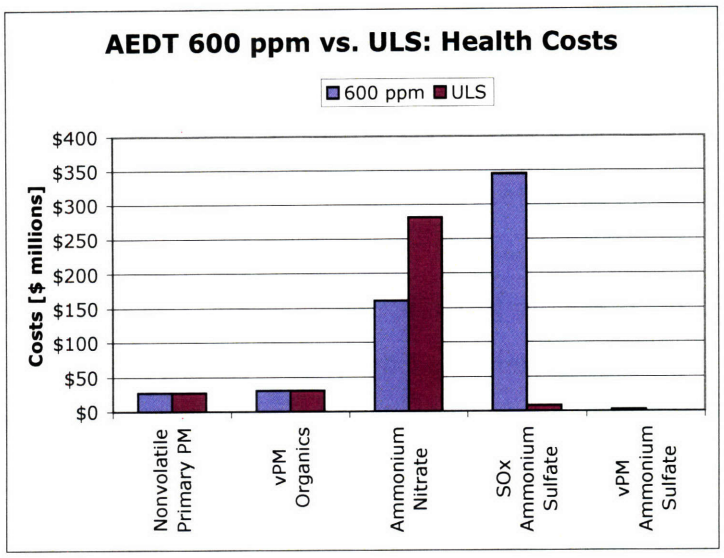

Figure 36: Health costs in the AEDT 600 ppm and ULS inventories

PM-related premature mortality incidences due to nonvolatile PM exposure remained approximately the same, at 5 incidences. Note that the hydrodesulfurization process used to create ULS fuel is likely to increase the fuel hydrogen content, ultimately reducing nonvolatile PM production, but this was not modeled in the inventory (Hileman 2007). It is possible that smoke number due to ULS fuel may be reduced by 4% in comparison with 600 ppm fuel. Also, the number of volatile primary PM-related deaths stayed approximately the same within the significant figures given.

Most importantly is that the number of ammonium sulfate-related premature mortality incidences due to aviation dropped drastically, from approximately 65 incidences ($345 million) to approximately 2 incidences ($9 million). The ammonium nitrate-related incidences increased due to the bounceback effect, from approximately 30 incidences ($161 million) to approximately 53 incidences ($281 million). This increase, however, was not enough to offset the decrease in ammonium sulfate-related incidences.

In total, the AEDT 600 ppm inventory led to approximately 110 incidences of premature mortality ($567 million), and the ULS inventory led to approximately 65 incidences ($349 million), a 38% reduction. The relative apportionment of these health incidences is shown in Figure 37 for the two inventories. In the ULS inventory, Los Angeles County

still had more incidences than any other county in the continental United States: approximately 10 incidences, or 9% of the total. This is a reduction from 20 incidences (19% of the total) in the 600 ppm AEDT inventory. The comparative apportionment for this county is shown in Figure 38.

There are several other factors that must be considered when analyzing the effects of ULS fuel. First, the EPA determined that creating diesel fuel with a 15 ppm sulfur content would reduce the volumetric energy content of the fuel by approximately 1.5 percent (U.S. Environmental Protection Agency 2000c), and a similar loss may occur for ULS aviation fuel (Massachusetts Institute of Technology and RAND Corporation 2007). The volumetric heat of combustion also drops by 1%, meaning that aircraft will need more fuel to fly a given distance in comparison with average Jet A. This ultimately translates to an increase in fuel prices for airlines.

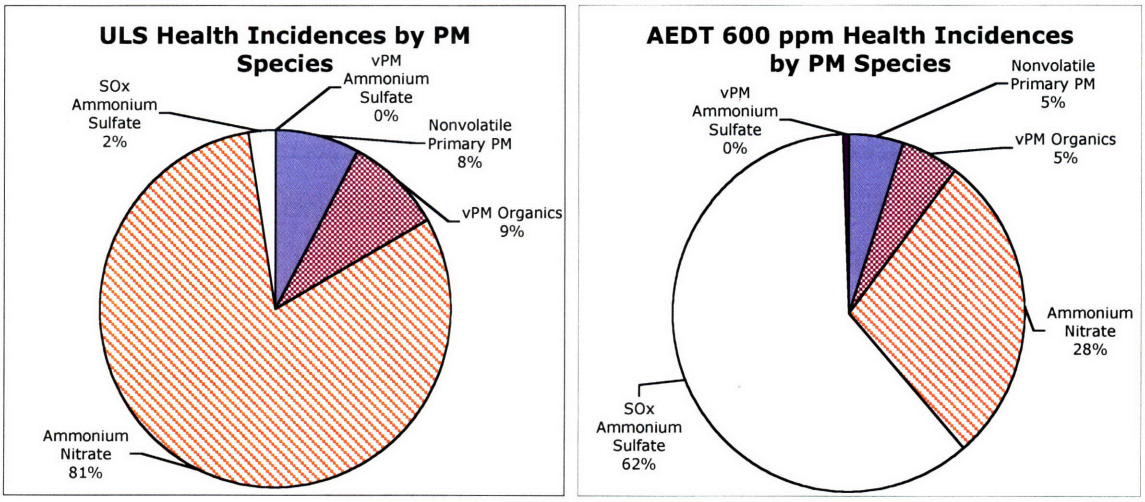

Figure 37: Relative apportionment of health incidences to PM concentrations for the AEDT 600 ppm and ULS fuel sulfur inventories

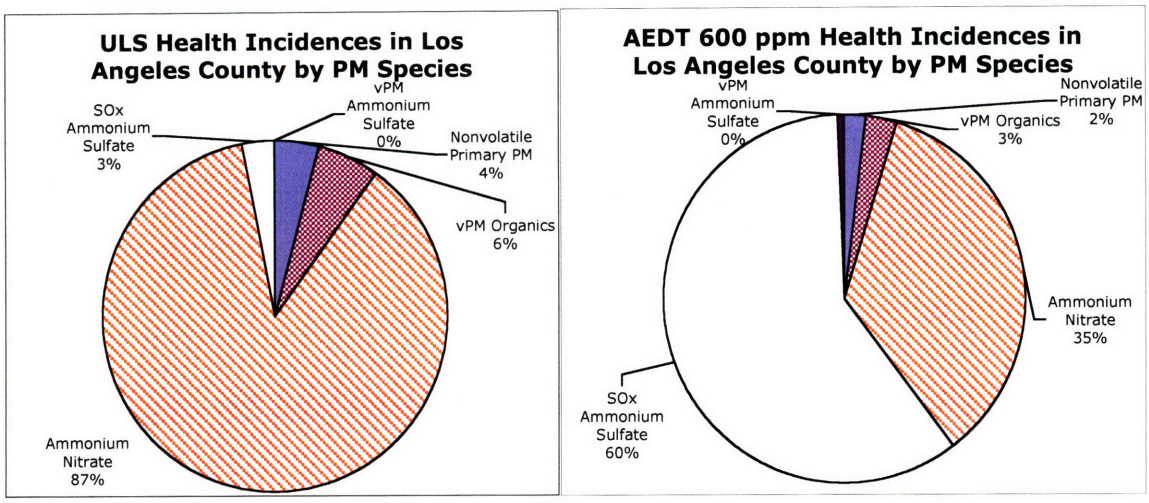

Figure 38: Relative apportionment of health incidences to PM concentrations in Los Angeles County for the AEDT 600 ppm and ULS inventories

Furthermore, there are costs that must be borne by fuel refineries to produce ULS fuel for aviation. EPA has estimated that a switch to ULS diesel fuel will cost approximately $50 million per refinery (U.S. Environmental Protection Agency 2000b). On a per-gallon basis, fuel costs will range from approximately 4 cents to 12 cents per gallon in various portions of the U.S due to costs in various processes such as the HDS process. HDS costs may be lower for ULS aviation fuel, however, as the molecules in aviation fuel have a lower boiling point.

The AEDT ULS inventory represented flights that burned approximately 2 billion gallons (6.7 megatonnes) of fuel in the U.S. in the year 2005. This represented an increase in fuel costs of approximately $260 million, assuming the upper bound of a 12-cent increase in costs to refineries per gallon burned. The total health costs avoided by an immediate switch to a ULS fuel (vs. using fuel with a sulfur content of 600 ppm) was $218 million, suggesting that the health benefits of a switch to ULS fuel may be comparable to the economic costs. Note that this cost-benefit analysis only takes into consideration refinery production costs and air quality-related health benefits due to reductions in PM exposure.

A switch to ULS fuels, however, also has implications for the global climate. The decrease in sulfate particles due to a switch to ULS fuel can cause a net warming of the

earth's climate, as sulfate PM has a cooling effect. The cost-benefit analysis in (Hileman 2007) using the Aviation Environmental Portfolio Management Tool (APMT) (Partnership for AiR Transportation Noise and Emissions Reduction 2007) indicated that the economic costs due to climate warming may outweigh the economic benefits due to improved air quality. Ultimately, policymakers must weigh these tradeoffs when considering a fuel switch.

3.3.3. Effects of a fuel sulfur stringency increase only for departures from Los Angeles County airports

Because Los Angeles County was responsible for the dominant portion of PM-related premature mortality in the Energy Policy Act study (20%), an analysis was done of a switch to ULS fuel only for departures out of airports in that county. All other counties in the analysis used 600 ppm fuel and had emissions as computed for the AEDT/SAGE 600 ppm inventory.

The results of the analysis are shown in Figure 39 and Figure 40. The switch to ULS fuel just for Los Angeles County reduced total PM-related deaths by approximately 10%, from 110 incidences ($567 million) to 100 incidences ($512 million). Los Angeles County was still responsible for 9% of the total health incidences. While the county still dominated in terms of total health incidences, air quality impacts due to aviation were substantially reduced in the county itself, reducing aviation-related mortality incidences by a factor of 2 (20 incidences to 10 incidences).

The cost to refineries to produce the ULS fuel for Los Angeles County was estimated to be approximately $10 million. The health benefits of using the ULS fuel were approximately $55 million, suggesting that ULS fuel only for departures out of Los Angeles County could be an effective strategy for mitigating premature mortality incidences due to PM exposure.

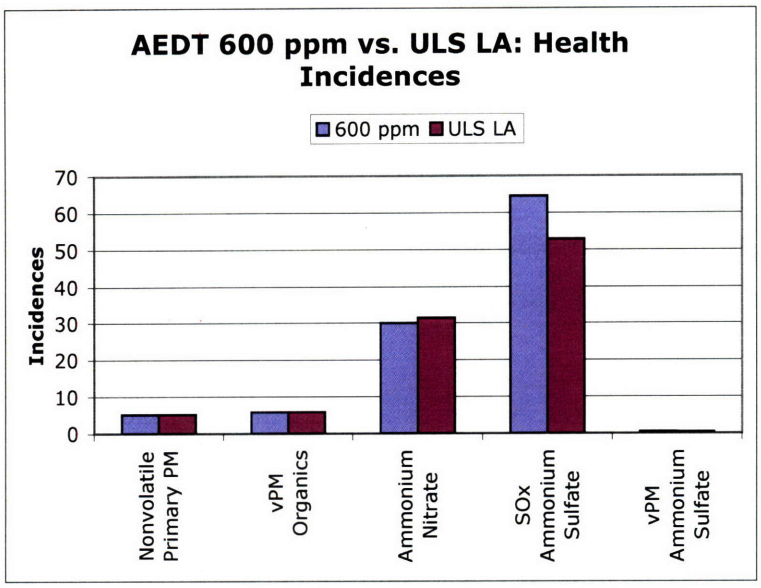

Figure 39: Comparison of the AEDT 600 ppm inventories and an inventory using ULS fuel for Los Angeles County airports only

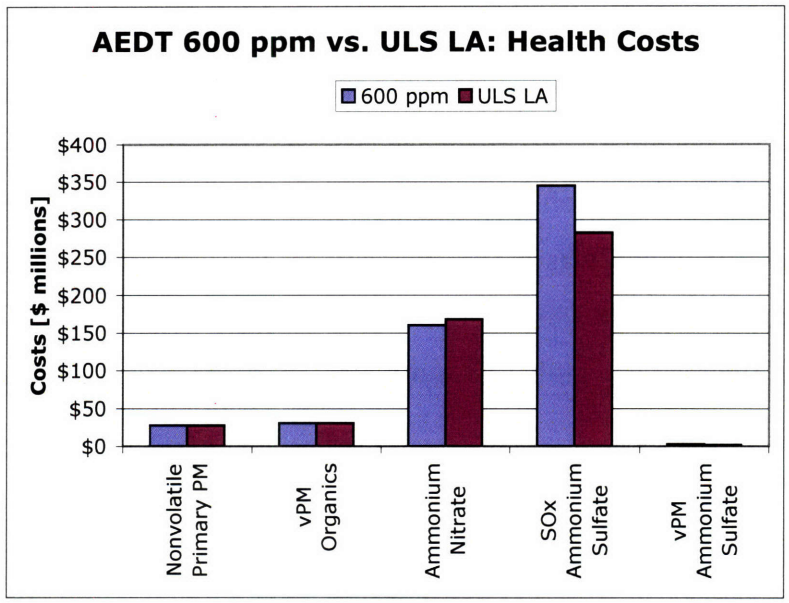

Figure 40: Health cost comparison between the 600 ppm inventory and an inventory with ULS fuel only for Los Angeles County

3.3.4. CAEP FESG assessment of a NOx stringency increase

The Forecasting and Economic Analysis Support Group (FESG) of ICAO's Committee on Aviation Environmental Protection performed an analysis of the economic effects of

NO_x stringency increases in 2004 (International Civil Aviation Organization 2004). Specifically, the group analyzed the ICAO NO_x regulations set at the 4th meeting of CAEP (known as the CAEP/4 regulations). CAEP/4 regulations are currently implemented in Section 2.3.2.c of ICAO Annex 16 Volume II (International Civil Aviation Organization 2005) and Section 87.21(d)(1)(vi) of 40 CFR 87 in the United States (Code of Federal Regulations 2005, Federal Register 2005).

FESG projected the effects of NO_x regulation changes to the year 2020 and the year 2030. To carry out the analysis, the group made several assumptions. In particular, they assumed that a new NO_x standard would take effect in a specific year (such as 2008 or 2012) and that the production of non-compliant engines would stop immediately at that point. The group also assumed that all demand for air travel could be met and that engine technology developments would not compromise payload capacity.

FESG estimated recurring and non-recurring costs to engine manufacturers as well as the costs of additional fuel, weight, landing fees, and maintenance for airlines. The group then computed the dollars of investment needed over the years to prevent a tonne of NO_x from being produced. Health effects were not explored. The analysis determined that the most cost-effective option was a 10% increase in NO_x stringency over CAEP/4 with implementation in the year 2008. This would lead to a cumulative NO_x reduction of 146,000 tonnes across the global aviation fleet in a 2002 – 2020 window, with no increase in CO_2.

The investment cost necessary to eliminate a tonne of NO_x was determined to be approximately $20,000 to $30,000 per tonne in 2002 dollars, with a 3% discount rate. While a direct comparison with the Energy Policy Act results is difficult, recall that the health cost of NO_x emissions per tonne was determined to be $2,000 in year 2001 dollars, with a 3% discount rate, in the Energy Policy Act study; see Section 1.4.6 of this thesis for further details.

3.4. Review

This chapter attempted to link aviation emissions regulations to the health effects of aviation's influences on air quality. Of the four ICAO standards (SN, NO_x, HCs, CO), the SN, NO_x and HC standards were deemed most influential to human health. The fuel sulfur standard set by ASTM international was deemed especially important, as ammonium sulfate particles could have been responsible for 46% to 69% of the human health effects due to PM exposure in 2005, and ammonium sulfate concentrations are related to the amount of sulfur in fuel. It was also shown that the NO_x standard continues to be important, as ammonium nitrate from NO_x was responsible for 18% to 20% of human health incidences due to PM exposure.

Concentrations of volatile primary PM from fuel organics were responsible for 6% to 18% of the health impacts, suggesting that a focus on hydrocarbon emissions is also important. It is known that HC emission indices increase sharply at low power settings. Nonvolatile primary PM concentrations caused 5% to 14% of PM-related mortality incidences, while volatile primary PM concentrations from sulfates caused 0% to 4%. The suggestion is that a focus on smoke number standards may not be as important as a focus on standards governing other emissions. Note that reducing fuel sulfur content will reduce ammonium sulfate concentrations from SO_x as well as from sulfate-related volatile primary PM.

Given that standards related to SO_x and NO_x were shown to be important, brief assessments were done of regulatory strategies relating to fuel sulfur as well as NO_x emissions. Using the AEDT/SAGE inventories, it was determined that a continental U.S.-wide move from fuel with 600 ppm sulfur content to fuel with 15 ppm sulfur content could reduce incidences of mortality by 38% from approximately 110 (using the 600 ppm inventory) to 65, avoiding $218 million in health costs due to premature mortality from PM. The potential cost to refineries of such a policy may be approximately $260 million, suggesting that the benefits and costs are comparable. It is important to note that the ammonium nitrate-related incidences in this scenario increased from 30 to approximately 53 due to the bounceback effect. It is also possible that

reducing sulfates will warm the climate because sulfate-related particles have a global cooling effect. When policymakers design regulatory strategies, they must be aware of tradeoffs such as these.

An analysis of the deployment of ULS fuel just for departures from Los Angeles County was also done. It was determined that such a deployment could reduce aviation-related incidences of premature mortality by 10% from 110 to 100 and avoid $55 million in health costs, with incidences in Los Angeles County being reduced by a factor of 2. It may cost refineries approximately $10 million to produce the ULS fuel, suggesting the efficacy of such a strategy. Finally, a brief assessment of ICAO FESG's exploration of an increase in NO_x stringency was performed. FESG estimated that the most cost-effective strategy for NO_x reduction would be a 10% increase in stringency in the year 2008, costing approximately $30,000 - $40,000 per tonne. However, the marginal damages of NO_x were found to be only $2,000 in the Energy Policy Act and RSM inventories.

Conclusion

Aviation is a critical part of the world economy, and the aviation industry is responsible for 5.4% of GDP in the United States (Joint Planning and Development Office 2007). Aviation activity also has quantifiable human health effects through perturbations to air quality. This thesis attempts to place aviation's air quality-related health impacts into the context of aviation standards specified in (International Civil Aviation Organization 2005), (Code of Federal Regulations 2005), and (ASTM International 2007). In order to evaluate the effectiveness of emissions standards, one must understand the pathway from regulations to health effects through emissions, dispersion, chemical transformation, and ultimately human exposure.

There are several questions of interest that have been addressed. The first is, "what are the local and regional air quality-related health impacts of aviation, and how do they relate to aviation emissions?" The results of a study mandated by Section 753 of the Energy Policy Act of 2005 were used to explore this question in conjunction with an analysis of the RSM inventory by CSSI, Inc. It was estimated that exposure to aviation-related particulate matter may cause between 140 and 160 yearly incidences of premature mortality of adults age 30 and up. Nonvolatile primary PM concentrations were responsible for 5% to 14% of this number. Volatile primary PM concentrations from fuel organics caused 6% to 18% of the incidences, while sulfate-related volatile primary PM concentrations caused 0% to 4% of the incidences. Secondary ammonium nitrate particles from NO_x emissions caused 18% to 20% of the mortality incidences, while ammonium sulfate particles from SO_x emissions caused 46% to 69% of the incidences.

The next question that this thesis sought to address is, "how do current aviation emissions regulations relate to the health impacts of aviation?" It was determined that health effects from exposure to nonvolatile primary PM emissions are related with the smoke number standard, and health effects from exposure to ammonium sulfate and sulfates-related volatile primary PM are strongly related to the standards that govern the amount of sulfur in fuel. Effects of exposure to ammonium nitrate are influenced by standards governing

NO_x emissions. It is known that hydrocarbon emission indices can be high at very low engine throttle settings, which suggests that the health effects of volatile PM from hydrocarbons relate to policies governing low-power activities such as aircraft idling and taxiing.

Finally, this thesis sought to explore the question, "what alternatives might policymakers consider for modifying the current regulatory strategies?" Regulatory strategies can involve the technology of aircraft and aviation fuel as well as aviation operations. Strategies have implementation costs, and it is important to compare these costs with the achievable benefits. This requires an awareness of the length of time needed for a strategy to become effective. Engine technology changes often require large investments of time and money for research, development, and manufacturing as well as the purchase of new engines and aircraft. For example, ICAO FESG's assessment of a NO_x stringency increase predicted that \$30,000 - \$40,000 of investment would be needed per tonne of NO_x eliminated for a 10% NO_x stringency increase in 2008, yet an assessment of the health effects of ammonium nitrate particles illustrated that the health cost of NO_x is approximately \$2,000 per tonne.

Operational and fuel technology changes may be done on a shorter time scale. Using the AEDT/SAGE 600 ppm and ULS inventories, an investigation of a continental U.S.-wide ULS fuel deployment found that mortality incidences related to aviation-influence PM concentrations were reduced from approximately 110 (\$567 million) to 65 (\$349 million), a reduction of \$218 million (38%). The additional cost to refineries to produce ULS fuel could be approximately \$260 million, suggesting that the costs and benefits of ULS fuel may be comparable. An exploration of ULS fuel only for takeoffs from Los Angeles County found that continental U.S.-wide mortality incidences were reduced by 10% from 110 (\$577 million) to 100 (\$512 million), with Los Angeles County deaths being reduced by a factor of 2. The costs to refineries to make this fuel could be approximately \$10 million, while the health benefits could be \$55 million. This may suggest that a ULS fuel policy is worth pursuing for Los Angeles County.

Policymakers must also consider possible costs and benefits that happen simultaneously in other areas. For example, reducing NO_x to improve air quality may increase CO_2, leading to climate warming. Similarly, the implementation of a ULS fuel strategy could have detrimental economic impacts due to climate change that outweigh the economic benefits of a reduction in PM-related health incidences because sulfate particles act to cool the climate. It is also important for policymakers to explore the various ways in which a regulatory strategy can affect public safety.

Recommendations for further research

Several areas of future research have been identified, and exploration of these areas is needed to more accurately assess the air quality-related health implications of aviation emissions regulations. First, it should be noted that the model deployed in this thesis used a linear scaling assumption to determine how changes in emissions inventories in counties with airports impacted PM concentrations in counties without airports. A comparison of the model with a higher-order model may be necessary to determine the quality of this assumption.

Next, improvements in knowledge and modeling of particulate matter chemistry and production are needed. As illustrated in this thesis, there are large differences between the FOA3 and FOA3a models. Also, it is still unclear how hydrocarbon emissions, which affect volatile PM concentrations, are related to engine and fuel technologies. Furthermore, research is needed to understand how the health effects of different PM species may differ. This thesis assumes that each PM species has the same health impact per unit mass inhaled by a population. Because PM models are a critical component of analyses of aviation's effects on human health, it is important to continuously enhance and refine the body of knowledge concerning PM.

Finally, researchers must continue to quantify the uncertainty in PM monitoring methods and technology as well as PM species apportionment methods and work to improve these areas. It is known that some PM species (particularly nitrates) are lost from FRM monitors, and this lost mass must currently be inferred. The mass of particle-bound water

changes based on ambient conditions and thus must also be inferred. The SANDWICH method assigns 12% of the inferred particle-bound water mass to nitrates, with the rest being assigned to sulfates. Changing this assignment to an upper bound value of 50% did not change the overall message of this thesis, but the quality of the apportionment assumption will become more important as policymakers and researchers desire analyses with finer resolution.

If aviation activity continues to grow over the next several decades, the importance of understanding aviation's health effects will only increase. Researchers play a critical role in building the body of knowledge related to how changing regulations ultimately changes human health effects. A better understanding of aviation-related particulate matter and the impact pathway from standards to health incidences can lead to better information for policymakers and better regulatory strategies.

Bibliography

Abt Associates Inc. 2006, *User's Manual for the Co-Benefits Risk Assessment (COBRA) Screening Model,* U.S. Environmental Protection Agency, Washington, DC.

Abt Associates Inc. 2005, Environmental Benefits Mapping and Analysis Program (BenMAP) User's Manual. Bethesda, MD, Abt Associates, Inc. Prepared for the U.S. Environmental Protection Agency Office of Air Quality and Standards.

AEA Energy & Environment 2007, *Heathrow Airport Emissions Summaries,* AEAT/ENV/R/2521 Final, AEA Energy & Environment, Warrington, UK.

ASTM International 2007, *Standard Specification for Aviation Turbine Fuels,* D1655-07, ASTM International, Committee D02 on Petroleum Products and Lubricants, PA.

BACK Aviation Solutions 2007, *BACK Website* [Homepage of BACK Aviation Solutions], [Online]. Available: http://www.backaviation.com/ [2008, 15 January].

Blevins, L.G. 2003, "Particulate matter emitted from aircraft engines", *AIAA Paper,* vol. 2764.

Brunelle-Yeung, E. 2007, *Change of Nonmelanoma Skin Cancer Incidence Related to Ozone Variation for Subsonic Fleet (DRAFT).*

Bureau of Transportation Statistics 2008, 15 January 2008-last update, *RITA | Bureau of Transportation Statistics (BTS)* [Homepage of Bureau of Transportation Statistics], [Online]. Available: http://www.bts.gov/ [2008, 15 January].

Chevron Corporation 2006, *Aviation Fuels Technical Review,* Chevron Global Aviation, Chevron Products Company, Houston, TX.

National Primary and Secondary Ambient Air Quality Standards. 40 CFR 50. 1 July 2007.

Reference Method for the Determination of Fine Particulate Matter as PM2.5 in the Atmosphere. 40 CFR 50 Appendix L. 2006.

Control of Air Pollution from Aircraft and Aircraft Engines. 40 CFR 87. 2005.

Reference Method for the Determination of Fine Particulate Matter as PM10 in the Atmosphere. 40 CFR 50 Appendix J. 1987.

CSSI Inc. 2006, *CSSI, Inc. - Moving Research to Reality.* Available: http://www.cssiinc.com/ [2008, 7 January].

Davidson, K., (Davidson.Ken@epamail.epa.gov), 27 July 2007, *Re: Ozone and PM health results*. Email to Christopher J. Sequeira.

Davidson, K., (Davidson.Ken@epamail.epa.gov), 17 November 2006a, *Fw: Configuration and APV Files*. Email to Christopher J. Sequeira.

Davidson, K., (Davidson.Ken@epamail.epa.gov), 17 November 2006b, *Fw: LocoMarine PM configs...* Email to Christopher J. Sequeira.

Dolwick, P., (Dolwick.Pat@epamail.epa.gov), 25 June 2007, *Re: EPAct: CMAQ SMAT'd PM2.5 outputs*. Email to Christopher J. Sequeira.

Dolwick, P., Kenneth Davidson and Christopher J. Sequeira, 25 September 2007, Phone call to Christopher J. Sequeira.

Dolwick, P. and Bryan Manning, 10 January 2008, *Discussion on National Emissions Inventory*. Phone conversation with Gayle Ratliff and Christopher J. Sequeira.

European Commission on Constrained Scenarios on Aviation and Emissions 2005, *CONSAVE 2050 - Constrained Scenarios on Aviation and Emissions,* European Commission on Constrained Scenarios on Aviation and Emissions. Available: http://www.dlr.de/consave/library.htm. [9 December 2007].

Federal Aviation Administration 2008, *Federal Aviation Administration* [Homepage of Federal Aviation Administration], [Online]. Available: http://aspm.faa.gov/main/atads.asp [2008, 15 January].

Federal Aviation Administration 2007a, 19 October 2007-last update, *ETMS* [Homepage of Federal Aviation Administration], [Online]. Available: http://hf.tc.faa.gov/projects/etms.htm [2008, 15 January].

Federal Aviation Administration 2007b, 4 October 2007-last update, *FAA - Aeronautical Informations Services* [Homepage of Federal Aviation Administration], [Online]. Available: http://www.faa.gov/about/office_org/headquarters_offices/ato/service_units/systemops/aaim/organizations/aeronautical_im/aero_info_svcs/nfdc/ [2008, 15 January].

Federal Aviation Administration 2006, 13 January-last update, *FAA - System for Assessing Aviation's Global Emissions (SAGE)* [Homepage of Federal Aviation Administration], [Online]. Available: http://www.faa.gov/about/office_org/headquarters_offices/aep/models/sage/ [2008, 10 January].

Federal Aviation Administration 2005, *Aviation & Emissions: A Primer,* Federal Aviation Administration, Office of Environment and Energy. Available: http://www.faa.gov/regulations_policies/policy_guidance/envir_policy/media/AEPRIMER.pdf. [August 24th 2007].

Control of Air Pollution From Aircraft and Aircraft Engines; Emission Standards and Test Procedures, Final Rule. 70 Fed. Reg. 69664-69687. 17 November 2005.

Control of Air Pollution From Aircraft and Aircraft Engines; Emission Standards and Test Procedures; Final and Proposed Rule. 62 Fed. Reg. 25356 - 25367. 8 May 1997.

Control of Air Pollution From Aircraft and Aircraft Engines; Emission Standards and Test Procedures. 47 Fed. Reg. 58462 - 58474. 30 December 1982.

Flagan, R.C. & Seinfeld, J.H. 1988, *Fundamentals of air pollution engineering,* Prentice Hall Englewood Cliffs, NJ.

Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D.W., Haywood, J., Lean, J., Lowe, D.C., Myhre, G., Nganga, J., Prinn, R., Raga, G., Schulz, M. & Van Dorland, R. 2007, "Changes in Atmospheric Constituents and in Radiative Forcing" in *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, eds. S. Solomon, D. Qin, M. Manning, et al., Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 130-234.

Frank, N.H. 2006, "Retained nitrate, hydrated sulfates, and carbonaceous mass in federal reference method fine particulate matter for six eastern U.S. cities", *Journal of the Air & Waste Management Association,* vol. 56, no. 4, pp. 500-511.

Greco, S.L., Wilson, A.M., Spengler, J.D. & Levy, J.I. 2007, "Spatial patterns of mobile source particulate matter emissions-to-exposure relationships across the United States", *Atmospheric Environment,* vol. 41, no. 5, pp. 1011-1025.

Harris, S.J. & Maricq, M.M. 2001, "Signature size distributions for diesel and gasoline engine exhaust particulate matter", *Journal of Aerosol Science,* vol. 32, pp. 749-764.

Health Council of the Netherlands 1999, *Public health impacts of large airports,* 1999/14E, Health Council of the Netherlands, Committee on the Health Impact of Large Airports, The Hague.

Hileman, J. 2007, *Fuel Sulphur Sample Problem Cost-Benefit Analysis (DRAFT),* CAEP-SG/20071-WP_XX, International Civil Aviation Organization, Committee on Aviation Environmental Protection (CAEP) Steering Group Meeting, 26 - 30 November 2007.

Intergovernmental Panel on Climate Change 2007, "Summary for Policymakers" in *Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, eds. M.L. Parry, O.F. Canziani, J.P. Palutikof, van der Linden, P.J. & C.E. Hanson, Cambridge University Press, Cambridge, UK, pp. 8-22.

International Civil Aviation Organization 2007a, *Annex 16 to the Convention on International Civil Aviation: Environmental Protection (Volumes I and II),* , International Civil Aviation Organization. Available: http://www.icao.int/eshop/pub/anx_info/an16_info_en.pdf. [20 November 2007].

International Civil Aviation Organization 2007b, *ICAO Environmental Report 2007,* , International Civil Aviation Organization, Committee on Aviation Environmental Protection, Montreal, Canada. Available: http://www.icao.int/icao/en/env/pubs/Env_Report_07.pdf. [4 November 2007].

Annex 16 to the Convention on International Civil Aviation, Volume II: Aircraft Engine Emissions, Amendment 5. 2005.

International Civil Aviation Organization 2004, *Economic Analysis of NOx Emissions Stringency Options,* CAEP/6-IP/13, International Civil Aviation Organization, Committee on Aviation Environmental Protection, Forecasting and Economic Analysis Support Group, Committee on Aviation Environmental Protection (CAEP) Sixth Meeting, Montreal, Canada, 2 - 12 February 2004.

Convention on International Civil Aviation Done at Chicago on the 7th Day of December 1944. Chicago, IL, 1944.

Joint Planning and Development Office 2007, *Business Case for the Next Generation Air Transportation System: Version 1.0,* , Joint Planning and Development Office. Available: http://www.jpdo.gov/library/nextgen_business_case_ver_1.pdf. [9 December 2007].

Kesselmeier, J. & Staudt, M. 1999, "Biogenic Volatile Organic Compounds (VOC): An Overview on Emission, Physiology and Ecology", *Journal of Atmospheric Chemistry,* vol. 33, no. 1, pp. 23-88.

Kugele, A., Jelinek, F. & Gaffal, R. 2005, *Aircraft Particulate Matter Emission Estimation through all Phases of Flight,* EEC/SEE/2005/0014, EUROCONTROL Experimental Centre, France.

Levy, J.I., Wilson, A.M., Evans, J.S. & Spengler, J.D. 2003, "Estimation of primary and secondary particulate matter intake fractions for power plants in Georgia", *Environmental science & technology,* vol. 37, no. 24, pp. 5528-5536.

Levy, J. and Ian Waitz, (iaw@mit.edu, jilevy@hsph.harvard.edu), 2 October 2007, *Fwd: revised text for aviation comparison.* Email to Christopher J. Sequeira.

Lukachko, S.P., Waitz, I.A., Miake-Lye, R.C. & Brown, R.C. 2005, "Engine Design and Operational Impacts on Particulate Matter Precursor Emissions", *ASME Turbo Expo GT2005-69112*Reno, NV, pp. 6.

Massachusetts Institute of Technology & RAND Corporation 2007, *Alternative Aviation Fuels Working Document - Version 1.2 (DRAFT),* MIT and RAND Corporation.

McNally, D. 2003, *Annual Application of MM5 for Calendar Year 2001,* , Alpine Geophysics, LLC, Arvada, CO.

Melissa Ohsfeldt, Ian Waitz, (mohsfeldt@cssiinc.com), 20 December 2007, *Fwd: EPACT issue with SOX.* Email to Christopher Sequeira and Elza Brunelle-Yeung.

Morrow, K., Hochard, D. & Francfort, J. 2007, *Cost Benefit Analysis Modeling Tool for Electric vs. ICE Airport Ground Support Equipment — Development and Results,* INL/EXT-07-12270, U.S. Department of Energy, Idaho National Laboratory, Idaho Falls, ID.

Ohsfeldt, M., Waitz, I., Sequeira, C., Thompson, T., Graham, M., Cointin, R., Gillette, W. & Gupta, M. 2007, "Quantifying the relationship between air traffic management inefficiency, fuel burn and air pollutant emissions", *7th FAA/EUROCONTROL ATM Seminar.*

Partnership for AiR Transportation Noise and Emissions Reduction 2007, *PARTNER - Project 3 / Valuation and Trade-offs of Policy Options* [Homepage of Partnership for Air Transportation Noise and Emissions Reduction], [Online]. Available: http://web.mit.edu/aeroastro/partner/projects/project3.html [2008, 11 January].

Penner, J.E. 1999, *Aviation and the Global Atmosphere: A Special Report of IPCC Working Groups I and III in Collaboration with the Scientific Assessment Panel to the Montreal Protocol on Substances that Deplete the Ozone Layer,* Cambridge University Press.

Pope, C.A., III, Burnett, R.T., Thun, M.J., Calle, E.E., Krewski, D., Ito, K. & Thurston, G.D. 2002, "Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution", *Journal of the American Medical Association,* vol. 287, no. 9, pp. 1132-1141.

Ratliff, G.L. 2007, *Preliminary Assessment of the Impact of Commercial Aircraft on Local Air Quality in the U.S.*, Massachusetts Institute of Technology, Engineering Systems Division, Technology and Policy Program.

Ratliff, G., Sequeira, C., Waitz, I., Ohsfeld, M., Thrasher, T., Graham, M., Thompson, T., Gillette, W., Gupta, M., Iovinelli, R., Holsclaw, C., Manning, B., Dolwick, P. & Davidson, K. 2008, *Aircraft Impacts on Local and Regional Air Quality in the United States: A Response to Section 753 of the Energy Policy Act of 2005 (DRAFT),* , Massachusetts Institute of Technology, Cambridge, MA.

Rojo, J.J. 2007, *Future Trends in Local Air Quality Impacts of Aviation,* Massachusetts Institute of Technology, Department of Aeronautics and Astronautics.

Smith, D.H., Malone, D.C., Lawson, K.A., Okamoto, L.J., Battista, C. & Saunders, W.B. 1997, "A National Estimate of the Economic Costs of Asthma", *American Journal of Respiratory and Critical Care Medicine,* vol. 156, no. 3 Pt 1, pp. 787-793.

Sorokin, A., Katragkou, E., Arnold, F., Busen, R. & Schumann, U. 2004, "Gaseous SO3 and H2SO4 in the exhaust of an aircraft gas turbine engine: measurements by CIMS and implications for fuel sulfur conversion to sulfur (VI) and conversion of SO3 to H2SO4", *Atmospheric Environment,* vol. 38, no. 3, pp. 449-456.

Timin, B., (Timin.Brian@epamail.epa.gov), 18 December 2007, *Re: Particle-bound water.* Email to Christopher J. Sequeira.

U.S. Environmental Protection Agency 2007a, *2007 Proposed Revisions to the National Ambient Air Quality Standards for Ground-Level Ozone: Chapter 6: Incremental Benefits of Attaining Alternative Ozone Standards Relative to the Current 8-hour Standard (0.08 ppm),* , U.S. Environmental Protection Agency. Available: http://www.epa.gov/ttn/ecas/regdata/RIAs/ozoneriachapter6.pdf. [9 December 2007].

U.S. Environmental Protection Agency 2007b, *Draft Regulatory Impact Analysis: Control of Emissions of Air Pollution from Locomotive Engines and Marine Compression-Ignition Engines Less than 30 Liters per Cylinder,* EPA420-D-07-001, U.S. Environmental Protection Agency, Office of Transportation and Air Quality. Available: http://www.epa.gov/otaq/regs/nonroad/420d07001.pdf. [12 January 2008].

U.S. Environmental Protection Agency 2007c, *Guidance on the Use of Models and Other Analyses for Demonstrating Attainment of Air Quality Goals for Ozone, PM2.5, and Regional Haze,* EPA-454/B-07-002, U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards, NC.

U.S. Environmental Protection Agency 2007d, *Integrated Science Assessment for Oxides of Nitrogen - Health Criteria: First External Review Draft,* EPA/600/R-07/093, U.S. Environmental Protection Agency, National Center for Environmental Assessment, NC.

U.S. Environmental Protection Agency 2007e, *Review of the National Ambient Air Quality Standards for Ozone: Policy Assessment of Scientific and Technical Information: OAQPS Staff Paper,* EPA-452/R-07-007, U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards, NC.

U.S. Environmental Protection Agency 2007f, *Technical Support Document: Modeling of the Impact of Aircraft Emissions on Air Quality in Nonattainment Areas (DRAFT),* EPA 454/R-07-xxx, U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards, NC.

U.S. Environmental Protection Agency 2006a, *2006 National Ambient Air Quality Standards for Particle Pollution: Chapter 5--Benefits,* U.S. Environmental Protection Agency. Available: http://www.epa.gov/ttn/ecas/regdata/RIAs/Chapter%205--Benefits.pdf. [18 November 2007].

U.S. Environmental Protection Agency 2006b, *Air Quality Criteria for Ozone and Related Photochemical Oxidants: Volume I,* EPA 600/R-05/004aF, Environmental Protection Agency, National Center for Environmental Assessment, NC.

U.S. Environmental Protection Agency 2005a, *Clean Air Interstate Rule Emissions Inventory Support Document,* U.S. Environmental Protection Agency. Available: http://www.epa.gov/cair/pdfs/finaltech01.pdf. [3 November 2007].

U.S. Environmental Protection Agency 2005b, *Regulatory Impact Analysis for the Final Clean Air Interstate Rule,* U.S. Environmental Protection Agency. Available: http://www.epa.gov/cair/pdfs/finaltech08.pdf. [18 November 2007].

U.S. Environmental Protection Agency 2005c, *Review of the National Ambient Air Quality Standards for Particulate Matter: Policy Assessment of Scientific and Technical Information: OAQPS Staff Paper,* EPA-452/R-05-005a, U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards, NC.

U.S. Environmental Protection Agency 2004a, *Air Quality Criteria for Particulate Matter: Volume I of II,* EPA/600/P-99/002aF, Environmental Protection Agency, National Center for Environmental Assessment, NC.

U.S. Environmental Protection Agency 2004b, *Air Quality Criteria for Particulate Matter: Volume II of II,* EPA/600/P-99/002bF, Environmental Protection Agency, National Center for Environmental Assessment, NC.

U.S. Environmental Protection Agency 2004c, *Clean Air Nonroad Diesel Rule Summary,* EPA420-F-04-029, U.S. Environmental Protection Agency. Available: http://www.epa.gov/nonroad-diesel/2004fr/420f04029.pdf. [8 December 2007].

U.S. Environmental Protection Agency 2004d, *Final Regulatory Analysis: Control of Emissions from Nonroad Diesel Engines,* EPA420-R-04-007, U.S. Environmental Protection Agency, Office of Transportation and Air Quality, Washington, DC.

U.S. Environmental Protection Agency 2002, *National Scale Air Toxics Assessment for 1996: Estimated Emissions, Concentrations and Risk: Technical Fact Sheet,* , Environmental Protection Agency, Office of Air Quality Planning and Standards.

U.S. Environmental Protection Agency 2000a, *Aircraft Contrails Factsheet,* EPA430-F-00-005, Environmental Protection Agency, Office of Air and Radiation, Washington, DC.

U.S. Environmental Protection Agency 2000b, *Clean Diesel Trucks, Buses, and Fuel: Heavy-Duty Engine and Vehicle Standards and Highway Diesel Fuel Sulfur Control Requirements, Chapter IV: Fuel Standard Feasibility,* EPA420-R-00-026, U.S. Environmental Protection Agency, Office of Transportation and Air Quality, Washington, DC. Available: http://www.epa.gov/otaq/highway-diesel/regs/ria-iv.pdf. [23 November 2007].

U.S. Environmental Protection Agency 2000c, *Final Regulatory Impact Analysis: Clean Diesel Trucks, Buses, and Fuel: Heavy-Duty Engine and Vehicle Standards and Highway Diesel Fuel Sulfur Control Requirements,* U.S. Environmental Protection Agency, Office of Transportation and Air Quality, Washington, DC. Available: http://www.epa.gov/otaq/highway-diesel/regs/2007-heavy-duty-highway.htm. [3 November 2007].

U.S. Environmental Protection Agency 2000d, *Guidelines for Preparing Economic Analyses,* EPA 240-R-00-003, U.S. Environmental Protection Agency, Office of the Administrator.

U.S. Environmental Protection Agency 1999, *Final Regulatory Analysis: Tier 2/Gasoline Sulfur Rule,* EPA420-R-99-023, U.S. Environmental Protection Agency, Office of Air and Radiation, Washington, DC.

U.S. Environmental Protection Agency 1994a, *Indoor Air Pollution: An Introduction for Health Professionals,* EPA 402-R-94-007, Environmental Protection Agency, Indoor Air Division, Washington, DC.

U.S. Environmental Protection Agency 1994b, *Review of the National Ambient Air Quality Standards for Sulfur Oxides: Assessment of Scientific and Technical Information: Supplement to the 1986 OAQPS Staff Paper Addendum,* EPA-452/R-94-013, Environmental Protection Agency, Office of Air Quality Planning and Standards, NC.

U.S. Environmental Protection Agency 1993, *Air Quality Criteria for Oxides of Nitrogen: Volume I,* EPA600/8-91/049aF, Environmental Protection Agency, Environmental Criteria and Assessment Office, NC.

U.S. Environmental Protection Agency 1986, *Review of the National Ambient Air Quality Standards for Sulfur Oxides: Assessment of Scientific and Technical Information: Addendum to the 1982 OAQPS Staff Paper,* EPA 450/05 86-013, Environmental Protection Agency, Office of Air Quality Planning and Standards, NC.

U.S. Environmental Protection Agency, National Oceanic and Atmospheric Administration 2007, 7 November 2007-last update, *Biogenic Emissions Inventory System (BEIS) Modeling | US DOC NOAA / US EPA* [Homepage of Environmental Protection Agency, National Oceanic and Atmospheric Administration], [Online]. Available: http://www.epa.gov/AMD/biogen.html [2008, 15 January] .

Air Quality Control Regions. 42 U.S.C. 7407. 3 January 2005a.

Aircraft Emission Standards. 42 U.S.C. 7571. 2005b.

Aircraft Emission Standards. 42 U.S.C. 7572. 2005c.

Aircraft Emission Standards. 42 U.S.C. 7573. 2005d.

Airport noise and access restriction review program. 49 U.S.C. 47524. 3 January 2005e.

Maintenance Plans. 42 U.S.C. 7505a. 3 January 2005f.

National Primary and Secondary Ambient Air Quality Standards. 42 U.S.C 7409. 3 January 2005g.

Energy Policy Act of 2005. P.L. 109-58. 2005.

Waitz, I., (iaw@mit.edu), 12 October 2007, *error in FOA3*. Email to Christopher J. Sequeira.

Wey, C.C., Anderson, B.E., Wey, C., Miake-Lye, R.C., Whitefield, P. & Howard, R. 2007, "Overview on the Aircraft Particle Emissions Experiment", *Journal of Propulsion and Power,* vol. 23, no. 5, pp. 898-905.

Yelvington, P.E., Herndon, S.C., Wormhoudt, J.C., Jayne, J.T., Miake-Lye, R.C., Knighton, W.B. & Wey, C. 2007, "Chemical Speciation of Hydrocarbon Emissions from a Commercial Aircraft Engine", *Journal of Propulsion and Power,* vol. 23, no. 5, pp. 912-917.

Appendix A: Valuation of health effects

The text below is duplicated from (U.S. Environmental Protection Agency 2007b) and refers to incidences due to exposure to particulate matter. The premature mortality endpoint also has been applied to ozone-related exposure for the 1990 and 2020 income levels in (U.S. Environmental Protection Agency 2007a).

Health Endpoint	Central Estimate of Value Per Statistical Incidence			Derivation of Estimates
	1990 Income Level	2020 Income Level[33]	2030 Income Level[34]	
Premature Mortality (Value of a Statistical Life): $PM_{2.5}$-related	$5,500,000	$6,600,000	$6,800,000	Point estimate is the mean of a normal distribution with a 95 percent confidence interval between $1 and $10 million. Confidence interval is based on two meta-analyses of the wage-risk VSL literature: $1 million represents the lower end of the interquartile range from the Mrozek and Taylor (2002)[35] meta-analysis and $10 million represents the upper end of the interquartile range from the Viscusi and Aldy (2003)[36] meta-analysis. The VSL represents the value of a small change in mortality risk aggregated over the affected population.

[33] Although the unit values presented in this table are in year 2000 dollars, all monetized annual benefit estimates associated with the proposed standards have been inflated to reflect values in year 2005 dollars. We use the Consumer Price Indexes to adjust both WTP- and COI-based benefits estimates to 2005 dollars from 2000 dollars (Council of Economic Advisors. 2005. The Annual Report of the Council of Economic Advisors. In: Economic Report of the President. Table B-60. U.S. Government Printing Office: Washington, DC.). For WTP-based estimates, we use an inflation factor of 1.13 based on the CPI-U for "all items." For COI-based estimates, we use an inflation factor of 1.24 based on the CPI-U for medical care.

[34] Our analysis accounts for expected growth in real income over time. Economic theory argues that WTP for most goods (such as environmental protection) will increase if real incomes increase. Benefits are therefore adjusted by multiplying the unadjusted benefits by the appropriate adjustment factor to account for income growth over time. For a complete discussion of how these adjustment factors were derived, we refer the reader to Chapter 9 of the CAND regulatory impact analysis (U.S. Environmental Protection Agency, 2004a. Final Regulatory Analysis: Control of Emissions from Nonroad Diesel Engines. EPA420-R-04-007. Prepared by Office of Air and Radiation. Available at http://www.epa.gov/nonroad-diesel/2004fr/420r04007.pdf). Note that similar adjustments do not exist for cost-of-illness-based unit values. For these, we apply the same unit value regardless of the future year of analysis.

[35] Mrozek, J.R., and L.O. Taylor. 2002. "What Determines the Value of Life? A Meta-Analysis." *Journal of Policy Analysis and Management* 21(2):253-270.

[36] Viscusi, V.K., and J.E. Aldy. 2003. "The Value of a Statistical Life: A Critical Review of Market Estimates Throughout the World." *Journal of Risk and Uncertainty* 27(1):5-76.

Health Endpoint	Central Estimate of Value Per Statistical Incidence			Derivation of Estimates
	1990 Income Level	2020 Income Level[33]	2030 Income Level[34]	
Chronic Bronchitis (CB)	$340,000	$420,000	$430,000	Point estimate is the mean of a generated distribution of WTP to avoid a case of pollution-related CB. WTP to avoid a case of pollution-related CB is derived by adjusting WTP (as described in Viscusi et al., [1991][37]) to avoid a severe case of CB for the difference in severity and taking into account the elasticity of WTP with respect to severity of CB.
Nonfatal Myocardial Infarction (heart attack) 3% discount rate Age 0–24 Age 25–44 Age 45–54 Age 55–65 Age 66 and over 7% discount rate Age 0–24 Age 25–44 Age 45–54 Age 55–65 Age 66 and over	 $66,902 $74,676 $78,834 $140,649 $66,902 $65,293 $73,149 $76,871 $132,214 $65,293	 $66,902 $74,676 $78,834 $140,649 $66,902 $65,293 $73,149 $76,871 $132,214 $65,293	 $66,902 $74,676 $78,834 $140,649 $66,902 $65,293 $73,149 $76,871 $132,214 $65,293	Age-specific cost-of-illness values reflect lost earnings and direct medical costs over a 5-year period following a nonfatal MI. Lost earnings estimates are based on Cropper and Krupnick (1990).[38] Direct medical costs are based on simple average of estimates from Russell et al. (1998)[39] and Wittels et al. (1990).[40] Lost earnings: Cropper and Krupnick (1990). Present discounted value of 5 years of lost earnings: age of onset: at 3%, at 7% 25-44: $8,774, $7,855 45-54: $12,932, $11,578 55-65: $74,746, $66,920 Direct medical expenses: An average of: 1. Wittels et al. (1990) ($102,658—no discounting) 2. Russell et al. (1998), 5-year period ($22,331 at 3% discount rate; $21,113 at 7% discount rate)
Hospital admissions				

[37] Viscusi, W.K., W.A. Magat, and J. Huber. 1991. "Pricing Environmental Health Risks: Survey Assessments of Risk-Risk and Risk-Dollar Trade-Offs for Chronic Bronchitis." *Journal of Environmental Economics and Management* 21:32-51.

[38] Cropper, M.L., and A.J. Krupnick. 1990. "The Social Costs of Chronic Heart and Lung Disease." Resources for the Future. Washington, DC. Discussion Paper QE 89-16-REV.

[39] Russell, M.W., D.M. Huse, S. Drowns, E.C. Hamel, and S.C. Hartz. 1998. "Direct Medical Costs of Coronary Artery Disease in the United States." *American Journal of Cardiology* 81(9):1110-1115.

[40] Wittels, E.H., J.W. Hay, and A.M. Gotto, Jr. 1990. "Medical Costs of Coronary Artery Disease in the United States." *American Journal of Cardiology* 65(7):432-440.

Health Endpoint	Central Estimate of Value Per Statistical Incidence			Derivation of Estimates
	1990 Income Level	2020 Income Level[33]	2030 Income Level[34]	
Chronic Obstructive Pulmonary Disease (COPD) (ICD codes 490-492, 494-496)	$12,378	$12,378	$12,378	The COI estimates (lost earnings plus direct medical costs) are based on ICD-9 code-level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total COPD category illnesses) reported in Agency for Healthcare Research and Quality (2000)[41] (www.ahrq.gov).
Pneumonia (ICD codes 480-487)	$14,693	$14,693	$14,693	The COI estimates (lost earnings plus direct medical costs) are based on ICD-9 code-level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total pneumonia category illnesses) reported in Agency for Healthcare Research and Quality (2000) (www.ahrq.gov).
Asthma Admissions	$6,634	$6,634	$6,634	The COI estimates (lost earnings plus direct medical costs) are based on ICD-9 code-level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total asthma category illnesses) reported in Agency for Healthcare Research and Quality (2000) (www.ahrq.gov).
All Cardiovascular (ICD codes 390-429)	$18,387	$18,387	$18,387	The COI estimates (lost earnings plus direct medical costs) are based on ICD-9 code-level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total cardiovascular category illnesses) reported in Agency for Healthcare Research and Quality (2000) (www.ahrq.gov).
Emergency Room Visits for Asthma	$286	$286	$286	Simple average of two unit COI values: (1) $311.55, from Smith et al. (1997)[42] and (2) $260.67, from Stanford et al. (1999).[43]
Respiratory Ailments Not Requiring Hospitalization				

[41] Agency for Healthcare Research and Quality (AHRQ). 2000. HCUPnet, Healthcare Cost and Utilization Project. Rockville, MD. http://www.ahrq.gov/HCUPnet/.

[42] Smith, D.H., D.C. Malone, K.A. Lawson, L.J. Okamoto, C. Battista, and W.B. Saunders. 1997. "A National Estimate of the Economic Costs of Asthma." *American Journal of Respiratory and Critical Care Medicine* 156(3 Pt 1):787-793.

[43] Stanford, R., T. McLaughlin, and L.J. Okamoto. 1999. "The Cost of Asthma in the Emergency Department and Hospital." *American Journal of Respiratory and Critical Care Medicine* 160(1):211-215.

Health Endpoint	Central Estimate of Value Per Statistical Incidence			Derivation of Estimates
	1990 Income Level	2020 Income Level[33]	2030 Income Level[34]	
Upper Respiratory Symptoms (URS)	$25	$27	$27	Combinations of the three symptoms for which WTP estimates are available that closely match those listed by Pope et al. result in seven different "symptom clusters," each describing a "type" of URS. A dollar value was derived for each type of URS, using mid-range estimates of WTP (IEc, 1994)[44] to avoid each symptom in the cluster and assuming additivity of WTPs. The dollar value for URS is the average of the dollar values for the seven different types of URS.
Lower Respiratory Symptoms (LRS)	$16	$17	$17	Combinations of the four symptoms for which WTP estimates are available that closely match those listed by Schwartz et al. result in 11 different "symptom clusters," each describing a "type" of LRS. A dollar value was derived for each type of LRS, using mid-range estimates of WTP (IEc, 1994) to avoid each symptom in the cluster and assuming additivity of WTPs. The dollar value for LRS is the average of the dollar values for the 11 different types of LRS.
Asthma Exacerbations	$42	$45	$45	Asthma exacerbations are valued at $42 per incidence, based on the mean of average WTP estimates for the four severity definitions of a "bad asthma day," described in Rowe and Chestnut (1986).[45] This study surveyed asthmatics to estimate WTP for avoidance of a "bad asthma day," as defined by the subjects. For purposes of valuation, an asthma attack is assumed to be equivalent to a day in which asthma is moderate or worse as reported in the Rowe and Chestnut (1986) study.
Acute Bronchitis	$360	$380	$390	Assumes a 6-day episode, with daily value equal to the average of low and high values for related respiratory symptoms recommended in Neumann et al. (1994).[46]

[44] Industrial Economics, Incorporated (IEc). March 31, 1994. Memorandum to Jim DeMocker, Office of Air and Radiation, Office of Policy Analysis and Review, U.S. Environmental Protection Agency.
[45] Rowe, R.D., and L.G. Chestnut. 1986. "Oxidants and Asthmatics in Los Angeles: A Benefits Analysis—Executive Summary." Prepared by Energy and Resource Consultants, Inc. Report to the U.S. Environmental Protection Agency, Office of Policy Analysis. EPA-230-09-86-018. Washington, DC.
[46] Neumann, J.E., M.T. Dickie, and R.E. Unsworth. March 31, 1994. "Linkage Between Health Effects Estimation and Morbidity Valuation in the Section 812 Analysis—Draft Valuation Document." Industrial Economics Incorporated (IEc) Memorandum to Jim DeMocker, U.S. Environmental Protection Agency, Office of Air and Radiation, Office of Policy Analysis and Review.

Health Endpoint	Central Estimate of Value Per Statistical Incidence			Derivation of Estimates
	1990 Income Level	2020 Income Level[33]	2030 Income Level[34]	
Restricted Activity and Work/School Loss Days				
Work Loss Days (WLDs)	Variable (national median =)			County-specific median annual wages divided by 50 (assuming 2 weeks of vacation) and then by 5—to get median daily wage. U.S. Year 2000 Census, compiled by Geolytics, Inc.
Minor Restricted Activity Days (MRADs)	$51	$54	$55	Median WTP estimate to avoid one MRAD from Tolley et al. (1986).[47]

[47] Tolley, G.S. et al. January 1986. *Valuation of Reductions in Human Health Symptoms and Risks. University of Chicago.* Final Report for the U.S. Environmental Protection Agency.